



PEDODIVERSITY

Editors

Juan José Ibáñez

James Bockheim

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PEDODIVERSITY

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PEDODIVERSITY

Editors

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Preface

The Earth surface is changing in an exponential way. These days our technological society and its demographical growth are impacting all natural resources with higher intensity than during the entire history of mankind. As a result of a plethora of perturbations (erosion, contamination, salinization, acidification, urban and industrial sealing, etc.) biosphere and the surface of the geosphere are being degraded or removed. Soil resources are vital to ensure the production of food to meet the world's increasing population as well as the support of life on the planet. However the pedosphere is subject to many risks and only in the last years has society begun to recognize that it must be preserved as part of our biological, geological and cultural heritage. Likewise soils are also an archive or memory of past climates, environments and ecosystems and thus if they are lost these archives of Earth history also will be lost.

Soil biodiversity may be greater than aboveground biodiversity; however, our understanding of soil system is incomplete. Pristine soils have begun to be rare natural bodies at the worldwide level. The lack of scientifically sound soil biodiversity inventories do not allow us to know the rate of extinctions of soil living organisms, particularly in view of their role in metabolizing and cycling nutrients and compounds in ecosystems. Therefore, soil cover has a vital role in biosphere biogeochemical cycles. Furthermore in view of our lack of knowledge of soil biodiversity, the only way to preserve organisms inhabiting in these natural bodies is to maintain their habitats as pristine as possible. Soil types or pedotaxa have been proposed in ecology as surrogate indicators of soil and aboveground biodiversity. In this framework, the design of soil reserves is a challenge that should be addressed (Ibáñez et al. 2003).

During the history of soil science, scientists and technicians have paid attention mainly to soil as the substrate for plant growth, sidestepping its environmental importance. This agronomic bias has hampered the investigation of soils and their function in many important environmental processes. Nowadays this agronomic focus is changing and recognition is being given that the pedosphere is an essential part of the climate or biogeospheric system. Thus the preservation of soil cover is essential for

the preservation of the biosphere. Furthermore the preservation of our geological heritage in general has begun to be recognized both by the scientific community and citizens.

While the role of biological diversity research has been recognized as one of the main topics of the ecological and conservation biology literature during the last six decades, an interest in soil diversity was not aroused until 1991 (Ibáñez et al. 1990). The first pedologist who tried to analyze and quantify pedological diversity of a territory was Russian, V.M. Fridland (1974, 1976). Regrettably his proposals and findings went unnoticed by the majority of his colleagues. At the beginning of the 1990s, Ibáñez and his coworkers applied mathematical tools developed by ecologists to study soil diversity (Ibáñez et al. 1990), or its neologism pedodiversity (McBratney 1992). Likewise the first paper which formally explained the main tools for the analysis of soil diversity or pedodiversity was five years later (Ibáñez et al. 1995, but see also the discussion paper of Ibáñez et al. in 1998). Since then, pedologists began to conduct research with the same framework and mathematical tools. Five years later pedodiversity analysis was recognized as an important pedometric tool (McBratney et al. 2000).

Pedodiversity analysis could be done using field information or digitalized soil surveys and data bases. Regrettably the lack of interest in updating the old soil information by policy makers of developed countries as well as the fact that in other parts of the world fundamental soil surveying, mapping and inventory had not been undertaken has hindered the progress of this subject. Natural resource classification schemes were a tool of the emerging discipline of pedodiversity. Some of these classification schemes are ad hoc and purpose-oriented and others attempt to be universal. For example, the USDA Soil Taxonomy and the WRB are being used as a language of communication as well as global inventories of natural-resource information. Thus pedodiversity determinations are taxonomic dependent.

This book compiles the main findings of 22 years of research in pedodiversity analysis. The authors are relevant authorities and part of the short history in this line of research. The chapters should be not considered as individual contributions but as a story told by its protagonists from their various perspectives along the same storyline. Thus this book, the first of its kind, is intended to be a combined handbook, an historical account of pedodiversity research and an essay on its future challenges.

A brief summary of the book is given. Chapter 1 discusses the short history of pedodiversity analysis showing its achievements and future challenges, as well as the conceptual and methodological concerns that affect this line of research. Ibáñez et al. demonstrate the striking similarities between the patterns of pedodiversity and biodiversity at different scales and environments. Finally these authors show evidence that the same

would happen with the diversity of other natural resources. They then discuss why an understanding of these similarities is important. Chapter 2 synthesizes the state of the art of the main mathematical tools used in diversity analysis, irrespective of the natural resource involved. In this chapter Enrico Feoli, Paola Ganis and Carlo Ricotta present the mathematical underpinnings of biodiversity and the utility of this approach to experts of other disciplines. Feoli and coworkers show the great difficulty to obtain scientifically sound statistical tools and algorithms that can satisfy all the practitioners of diversity research, as well as the unlikelihood that a few set of procedures will become available to solve all scientific and social demands about how diversity should be measured. Chapter 3 deals with the relationships between pedodiversity and non-linear dynamics, showing that the trajectories of soil systems and soilscares through time are difficult to predict as a result of the intrinsic and extrinsic factors that affect them, as also occurs with biodiversity of ecosystems. In his chapter Jonathan Phillips describes pedodiversity analysis as a new perspective for understanding the genesis of soils and the underlying regolith. Likewise, this author defends, like others in this book, that the increase or decrease of soil diversity is in part the result of the above mentioned non-linearity. Where nested non-linear systems have different properties at different scales of its hierarchy, other features are scale-invariant. In Chapter 4, Fernando San-José and Javier Caniego explain the origin and nature of scale-invariant structures and processes of environmental resources in general, paying special attention to soils. After that these authors identify intriguing relations between fractals, multifractals and diversity. Finally they detect the scale-invariance similarities that appear between biodiversity and pedodiversity patterns.

Chapter 5 contextualizes the role of pedodiversity and biodiversity in the framework of landscape-ecology science, showing once again the strong similarities between biological and pedological assemblages. Finally Asunción Saldaña addresses the importance of the design of networks of soil reserves to preserve soils as part our geological, biological and cultural heritages. In Chapter 6 Norair Toomanian analyzes the relations between soils and landforms as well as their respective diversities. As with pedodiversity-biodiversity relationships, the detected patterns in landform diversity seems to follow the same mathematical patterns, suggesting that the structure, dynamics and evolution of these natural resources (but see also litho-diversity or the variety of rock types) could be interrelated.

The human impact on the pedosphere is a cause for concern, in view of the fact that pristine soils are becoming rare, and most pedotaxa are either perturbed or domesticated. For this reason, some soil types or pedotaxa warrant preservation. Chapter 7 raises the importance of soil endemism in pedodiversity and pedogeography analysis. The authors, James G. Bockheim and Nicholas Haus, discuss the concept of soil endemism, procedures to

detect endemic soils, the relation between soil endemism and soil richness, the importance of protection of endemic soils from different points of view (natural and cultural heritage, blocks of memory of past environments, climates, etc.) and the relationship between soil and plant endemism. The authors suggest that soil endemism can be used as a management tool in protecting rare and endangered soils and in identifying sites for specific land-use practices.

As stated above, the pedosphere and many soil types are at risk of extinction as result of intensive and widespread impact of human activities. Land degradation, industrial farming and urban sprawl are the main driving forces of pedodiversity changes/losses. Chapter 8 deals with the terrifying impact of the industrial agriculture on pedodiversity and soil assemblages, showing the extinction of some pedotaxa as well as the emergence of new manmade soils that otherwise might not occur in nature. In this chapter Giuseppe Lo Papa and Carmelo Dazzi analyze the structure and evolution of the soilscape, showing the heavy impact of some aggressive technological land management practices over the past 40 yr. These authors make use of sophisticated mathematical tools, such as cellular automata and Markov modelling, to predict pedodiversity loss in their study area for the year 2050. In contrast, in the most industrial areas of the planet, soil loss and potential pedodiversity changes occur mainly as a result of the exponential growth of cities and human infrastructures. In Chapter 9 several representative examples are analyzed in the most populated and developed provinces of China. The results obtained are a matter of concern. However in this chapter Zhang Xuelei goes beyond by synthesizing the state of the art of pedodiversity researches in China. Many papers on this topic were written in Chinese and thus the results obtained in general have not been read by experts in the western world. The effort of synthesis made by Zhang Xuelei will allow readers to understand the progress made in this Asiatic country on pedodiversity analysis. His studies confirm that most of the trends identified in the western world have been explained in previous chapters of this book.

Finally the editors synthesize the contents of the above mentioned chapters written in this book in a set of short conclusions. It has been an easy task in view that the same patterns have been detected by distinct experts in different territories and at different scales, against a certain methodological chaos and the lack of consensual standards and protocols to measure soil diversity. A doctrinal corpus begins to emerge in pedodiversity analysis. However, as yet much remains to be done in order that we can speak of a new branch of the pedology.

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CHAPTER 1

Pedodiversity State of the Art and Future Challenges

Juan José Ibáñez,^{1,*} Ronald Job Vargas² and
Antonio Vázquez-Hoehne³

1. Introduction: The Dark Foundation of Many Natural-resource Sciences

At first, the notion of diversity seems clear and intuitive. However, other terms have sometimes been used synonymously, such as variety, heterogeneity, variability, complexity, etc. At the scientific level these synonymous usages cause confusion and vagueness. Since the foundation of natural-resource sciences, experts from different disciplines have recognized the impressive global diversity surrounding them, regardless of whether their interest was in living organisms, rocks, landforms or soils. The increased diversities over larger spaces and areas can probably be traced back centuries, perhaps including prehistoric times, when firsthand knowledge of natural resources and their diversity was fundamental for survival. Thus, any search for the foundation of the diversity concept in any branch of natural sciences must be subjective, biased and the origin somewhat arbitrary.

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This search must start with the operational definition of the concept of diversity. How should biological diversity, soil diversity, etc. be defined? How must it be measured? What are the purposes of such studies? Unfortunately, many researchers confuse the founders of a scientific discipline with later experts who simply coined the term. Coining a neologism is not a scientifically founding work. Thus, for example neologisms such as biodiversity and pedodiversity were proposed after systematic scientific biological and soil diversity studies began. Historical inquiries become more confusing, yet there may be several synonyms for the same concept. Controversies and disputes may also arise, regrettably, due to personal vanities and national and disciplinary bias (Fig. 1.1).

As far as we know, the first scientist to show serious interest in the development of the concept of soil diversity was the Russian pedologist V.M. Fridland (1974, 1976). Regrettably, many interesting ideas proposed by this notable researcher have been cited in scientific literature, but have not been implemented in practice (Hole and Campbell 1985). Diversity analysis was not an exception (McBratney 1995, Ibáñez 1996).

Diversity indices based on information theory (Shannon and Weaver 1949) have been utilized in many sciences for a wide variety of purposes. The same occurred in pedology. For example, Minasny et al. (2010) wrote that Jacuchno (1976) and Linkeš et al. (1983) made use of Shannon's entropy

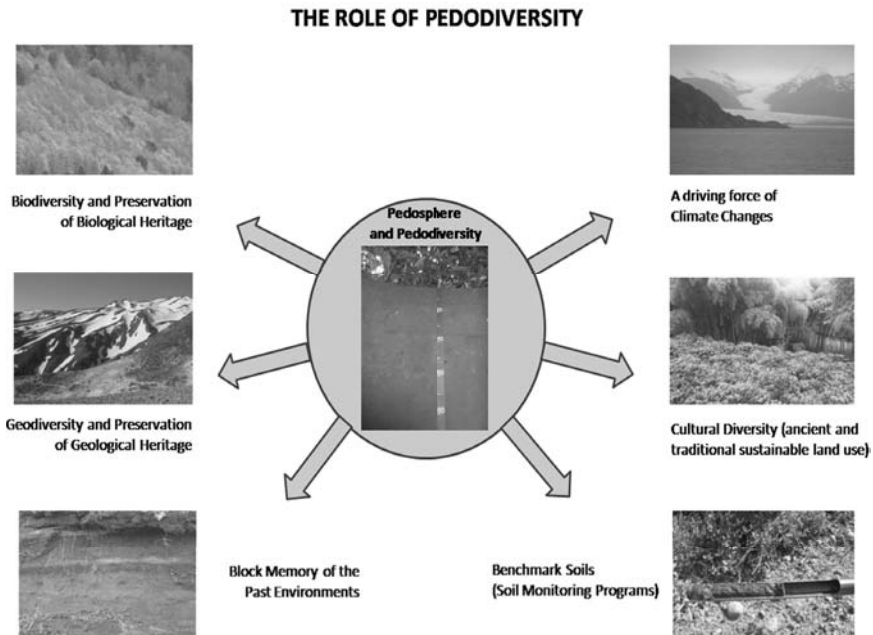


Figure 1.1 Six reasons for preservation of a pedological heritage.

and evenness for an evaluation of soil cover heterogeneity in Russia and Slovakia, respectively. However the purpose of these authors was the selection of extensive homogeneous field areas for agricultural purposes, rather than pedodiversity analysis as it is known these days. Therefore these scientists were not involved in diversity studies directly, thus they should not be considered as pioneers of the later line of research. Likewise Beckett and Bie (1978), in a very interesting report on soil survey map analysis, found that the number of soil types and soil series in Australia depend on the area surveyed. They used log-log plots to show that pedotaxa-area relations conform to a power law. However the purpose of their study was not a soil diversity analysis per se; rather, it was a review of soil survey procedures and standards. Thus, it is debatable whether these authors should be included among the founders of the study of pedodiversity. As has already been indicated, the perception about the increment of number of taxa related to the area surveyed was something intuitive for naturalists.

Recognizing that Fridland was the first pedologist to operationalize the notion of soil diversity, Ibáñez et al. (1990a) introduced the mathematical tools (richness, Shannon diversity index, Shannon equitability) used in ecological diversity for studying spatial soil patterns. After the Rio Summit in 1992 and following the impact generated in public opinion by the neologism of biodiversity proposed by Wilson and Peter (1988), the Australian pedometrician Alex McBratney coined the neologism of pedodiversity (McBratney 1992), whereas the Australian geologist Sharples (1993) proposed geodiversity.

2. What is Diversity?

A countless number of definitions have been proposed for biodiversity in the scientific literature. However, this does not apply to the definition of pedodiversity. The clearest and most neutral definition of diversity is that of Huston (1994):

*“The concept of diversity has two primary components, and two unavoidable value judgements. The primary components are statistical properties that are common to any mixture of different objects, whether the objects are balls of different colours, segments of DNA that code for different proteins, species or higher taxonomic levels, or **soil types** or habitat patches on a landscape. Each of these groups of items has two fundamental properties: 1. the number of different types of objects (e.g., species, **soil types**) in the mixture or sample; and 2. the relative number or amount of each different type of object. The value judgements are: 1. whether the selected classes are different enough to be considered separate types of objects; and 2. whether the objects in a particular class are similar*

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enough to be considered the same type. On these distinctions hangs the quantification of biological diversity”.

Thus, there are essentially two different components in the concept of diversity: the variety of species (richness), and the way in which the individuals are distributed among those species (evenness or equability). Indices of diversity either attempt to incorporate both components of diversity into a single value, or they tend to neglect one or the other component. From a methodological point of view, the most popular methods for diversity analysis may be grouped into two general classes (Magurran 1988):

1. Indices of richness: number of different objects on the site (i.e., account of the number of biological species or soil types known to occur in a defined sampling unit).
2. Object abundance models: a model of distribution which provides the closest fit to the observed pattern of object's abundance (e.g., geometric series, log series, lognormal distribution, broken stick model, etc.).

However, some authors consider that taxa differ in the degree of similarity-dissimilarity and suggest that this taxonomic distance should be included (e.g., Rao 1982, McBratney and Minasny 2007). As we can see in Section 5.1, this alternative has pros and cons.

The most popular index in diversity analysis are S , $SHDI$ (or H') and $SHEI$ (or E). The Richness index (S) is simply the number of different soil types, corresponding to the number of soil to a specific taxonomic system (i.e., soil subgroups). The Shannon's Diversity Index ($SHDI$) and the Shannon's Evenness Index ($SHEI$) are calculated as follows:

$$SHDI = -\sum_{i=1}^n p_i \times \ln p_i$$

where p is the portion of soilscape occupied by a soil class i , and n is the total number of soil classes forming the soilscape. It is one of the most popular measures of diversity in community ecology, applied here to soilscares. $SHDI$ ranges > 0 with no limits and equals 0 when the soilscape contains only one class (i.e., no diversity).

The Shannon's Evenness Index is calculated as follows:

$$SHEI = \frac{-\sum_{i=1}^n p_i \times \ln p_i}{\ln n}$$

where p is the portion of soilscape occupied by a soil class i , and n is the total number of soil classes forming the soilscape. It expresses conceptually

the complement of dominance. An even distribution of area among class types results in maximum evenness. SHEI ranges between 0 and 1: it is 0 when the soilscape contains only one class (i.e., no diversity) and approaches 0 when the distribution of area among the different classes becomes uneven (i.e., dominated only by one class type). It equals 1 when the distribution of area among classes is perfectly even (i.e., proportional abundances are the same).

3. The Dilemma of a World Full of Diversities

As with ecologists, pedologists often consider that taxonomic diversity (diversity of soil types or pedotaxa) should not be the only objective of a pedological investigation. Thus, some experts analyze the genetic diversity (soil horizons assembled in pedons), functional diversity (soil functions), etc. (e.g., Ibáñez 1996, Ibáñez et al. 1998, Ibáñez and De-Alba 1999, Odeh 1998, Toomanian and Esfandiarpour 2010). However, to date only taxonomic and genetic pedodiversity have been analyzed (e.g., Saldaña and Ibáñez 2007, Toomanian and Esfandiarpour 2010). A clearer understanding of functional diversity is needed. A plethora of concepts and different mathematical tools to formalize each of them could produce confusion and vagueness, as it occurred some time ago in biodiversity analysis (e.g., Peters 1991, Ricotta 2005). A dilemma appears from this point of view. International scientific journals prioritize methodological papers, publishing a plethora of indices and approaches. The search for patterns at different scales and environments can only be done if the researchers make use of the same group of concepts and mathematical tools. Thus, the last approach is hampered by the former, preventing the detection of universal regularities and the growth of a scientifically sound pedodiversity theory (see also Section 6.8). The search for new mathematical tools may be detrimental to the search for universal soil patterns and may not represent a scientifically sound solution. Both approaches should be balanced.

4. The Various Purposes of Pedodiversity Analysis

Pedodiversity tools could be used for different purposes. In our opinion, the most important are the identification of the following:

- Diversity patterns of pedological assemblages (soilscares, soil regions, etc.);
- Pedogenetic diversity (diversity of genetic soil horizons in a given territory);
- Soil richness and pedodiversity increases with the sampled area (pedodiversity-area relationships);

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- Soil richness and pedodiversity increases or decreases with time in chronosequences such as islands of different age in a given archipelago, fluvial and marine terraces, etc. (diversity-time relationships);
- Other regularities in soil assemblages such as potential nesting among them, species-range size distribution, scale invariance or scale dependence of soilscape patterns etc.;
- Areas for designing networks of soil reserves;
- Diversity patterns in space and time of different natural resources (e.g., soils, rocks, landforms, biological diversity, etc.);
- Quantitative mathematical concepts and tools in soil geography (e.g., quantification of soil endemisms, soil minorities, as well as corroboration/refutation of older concepts such as the zonal paradigm);
- Controls of the nature of soils and soilscape assemblages (e.g., are there dissipative structures, nonlinear or complex systems?);
- Spatial representation of the soil properties variability across the landscapes using soil taxonomies.

Some of these items are related, and mixing them will possibly lead to new findings. For example, to date, most pedogenetic studies have been concerned mainly with soil formation over time. However, pedologists have shown much less interest in pedogenesis from a spatial perspective (e.g., soilscares using digital mapping tools). In chapter 3, Jonathan Philips analyzes this topic in depth, showing that pedodiversity tools open new perspectives on pedogenetic theory, refuting some old concepts.

5. Methodological Aspects

5.1 Classifications and diversity

Diversity can be analyzed in any context where it is possible to establish a classification or taxonomy. Because universal classifications aim to address the global diversity of natural resources, they should be recognized as global inventories. Some pedologists disregard the use of classical taxonomies in soil science, claiming that “ad hoc” classifications, i.e., those classifications for specific purposes usually using numerical tools ones, are better (e.g., Odeh 1998). However, Phillips and Marion (2007) and Petersen et al. (2010) tested both approaches and reported that only minor differences exist when classical and “ad hoc” classifications have been compared. Likewise McBratney and Minasny (2007) argue that the incorporation of taxonomic distance (taxonomic differences between soil classes) in a diversity index should improve the estimation of pedodiversity. However, Toomanian

and Esfandiarpour (2010) demonstrated that in only a few cases minor differences have been detected.

For practical purposes, it would be advisable to make use of “ad hoc” classifications. However, if we are interested in the comparison of pedodiversity studies, widely used taxonomies are strongly recommended (Ibáñez and Saldaña 2008, Ibáñez et al. 2008). So, we examine the classical categorization of soils in the same way as standard biotic components of ecosystems are analyzed in biodiversity studies. Actually, the objects belonging to the same taxonomic group can be counted or the spatial extent of each pedotaxa in the area of study evaluated (e.g., Magurran 1988). The International Union of Soil Sciences (IUSS) recognized the World Reference Base for Soil Resources (IUSS Working Group WRB 2006) as the world classification system, although the USDA Soil taxonomy (Soil Survey Staff 2010) is also applied by many pedologists worldwide.

Other types of pedodiversity indices without the use of both classical and “ad hoc” classification systems have been proposed. Petersen et al. (2010) measured the variable space (VS): calculated from the n-dimensional space of normalized soil properties. Thus, as these authors remark: *“The resulting values are not numbers of soils like calculated in the taxonomic and parametric methods, but n-dimensional hypervolumes that include all the observations”*. Although it is an interesting metric, VS does not correspond with the notion of diversity and variety in terms of discrete objects (taxa). This dilemma brings us back to the conflict in the use of synonymous notions of richness, diversity, variety, heterogeneity, variability, complexity, etc. Therefore, there should be a consensus or at least clarification of the terminology to avoid confusion and enable comparative analysis. For example, in scientific literature, diversity goes beyond richness and takes into account relative abundance. Similarly, the consideration of the interrelations among taxa (connectivity) leads to the concept of complexity. Ibáñez and De-Alba (2001) and Saldaña and Ibáñez (2007) outlined the concepts of connectance and complexity in the literature on pedodiversity similar to what has been done for decades in the ecological literature. However this dilemma is far from being resolved, and additional efforts will be required.

In the debate on the best pedodiversity index, recently several pedologists have argued that taxonomic distance must be included jointly with pedotaxa richness (pedorichness) and the respective abundances of each soil type (e.g., McBratney and Minasny 2007, Toomanian and Esfandiarpour 2010, Petersen et al. 2010), particularly through the use of the Rao quadratic entropy index “Q” (Rao 1982). However, Ricotta (2005) summarized the state of the art of this index in biodiversity analysis, showing that it violates at least two normally undisputed axioms: (i) for a given number of species N, maximal diversity arises for equi-probable species distribution and (ii) the permutation invariance (Pielou 1975). The last axiom postulates that

diversity values corresponding to the relative abundances p_1, p_2, \dots, p_N and to a p'_1, p'_2, \dots, p'_N permutations of them are identical (see Ricotta 2005 and references therein). Thus the Q index is not a scientifically sound solution to take account of taxonomic distance in a single diversity index and falls into the category “weak diversity index” (Ricotta 2005). In addition, a scientifically sound measure of the taxonomic distinctiveness is also a difficult task (Ricotta 2005). In view of this, it is debatable whether it is preferable to add taxonomic distance to previous diversity algorithms in deriving a new one or measure taxonomic distinctiveness and richness, excluding abundance in other index independently (Vane-Wright et al. 1991). As we can see in Section 8, the plethora of proposed indices has more cons than pros to the progress of this line of research.

5.2 Diversity, taxonomies, scales and sampling intensity

Before displaying the most relevant patterns found to date in the literature of pedodiversity, several questions and considerations must be taken into account. As in biodiversity studies (e.g., Ricotta 2005), pedorichness and pedodiversity values are dependent on: (i) taxonomy; (ii) scale; and (iii) sampling intensity. The number of pedotaxa of a given soil taxonomy, as well as the hierarchical level used, determine the figures obtained. It is obvious that in hierarchical classifications the number of taxa increases from the top to the bottom and consequently pedodiversity values depend on the hierarchical level used. It is an intrinsic problem inherent to diversity analysis, independent of the natural resource involved. Likewise, because the number of taxa may increase over short distances as result of divergent pedogenesis (see Section 6.8, and Phillips in this book) the design and intensity of the sampling scheme (Fig. 1.2) and the skills of the surveyor determines the number of soil types that finally will be represented in maps. Finally, soil diversity estimates could be based on previously nested soils maps or georeferenced data bases. As a consequence of map generalization using classical procedures on choropleth maps, considerable information is lost at small scales; in contrast, digital soil mapping technologies avoid these constraints. Therefore, results obtained on non-nested small scale soil maps can be considered only as rough estimates. Ibáñez et al. (1995) and Hupy et al. (2004) show two examples on the repercussions of the map scale on pedodiversity values. Figure 1.3 shows the loss of information according to a decrease in scale from detailed to coarse maps. All these items must be borne in mind when assessing comparability between different studies. For these reasons, the detected trends sometimes will be more important than mathematical indices per se.

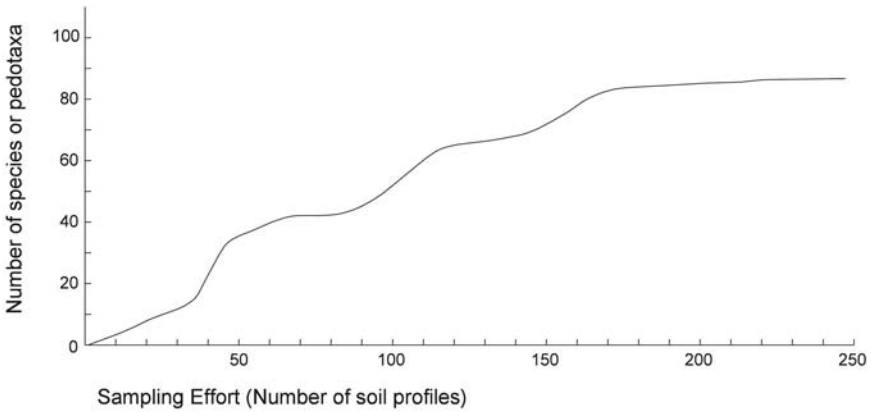


Figure 1.2 Pedodiversity and sampling effort: cumulative taxa number.

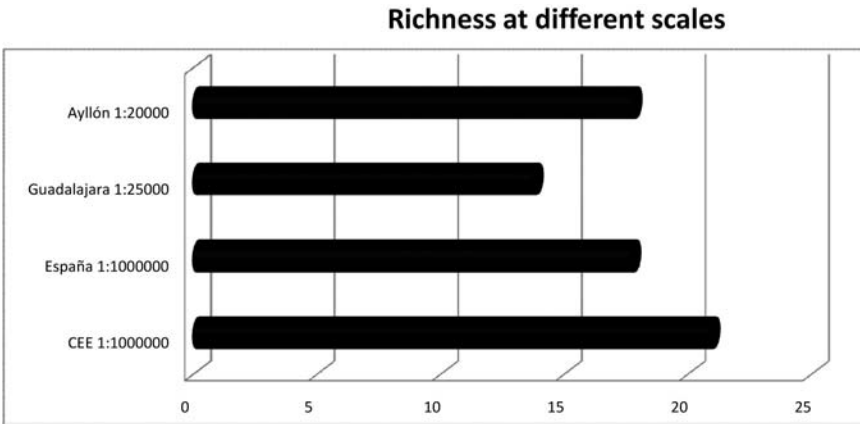


Figure 1.3 Scale dependence of pedodiversity analysis: richness of major soil groups (FAO 1971) for soil maps, the areas and scales of which differ by orders of magnitude and different sampling efforts (after Ibáñez et al. 1995).

6. Patterns Detected in Pedodiversity Analysis

6.1 The strange asymmetric distributions of the natural bodies in nature: the ubiquity of the Willis curve

An intriguing fact is the ubiquity of the so-called hollow or Willis curve in all inventories of all natural resources inventoried (Ibáñez et al. 2005a,b). The ubiquity of this curve is exemplified within essentially all groups of organisms. The origin of this term comes from the concave shape of frequency distribution of the taxonomic assemblages when its taxa are

ranked from the most to the least abundant one (Fig. 1.4). The same is true for the frequency distribution of taxonomic subunits per unit, when units are ordered from the most to the least abundant ones (subtaxa/per taxa using a given distribution). Thus, the Willis curve shows that there are many rare taxa within an assemblage and just few very abundant taxa. Willis and Yule (1922) reported that hollow curve distributions are well fitted to a power law or related statistical distribution models. All biodiversity and pedodiversity inventories among others follow this trend, as also happens with the structural analyses of biological and pedological taxonomies (Ibáñez et al. 2006). However, the origin and causation of the Willis curves are unknown. In diversity algorithm terms this feature means that the evenness index (“*E*”) will not get the highest possible figure, and thus, the Shannon entropy index (“*H*”) for a given number of species (“*S*”) never reaches its potential maximum value (all the taxa have a equiprobable number of individuals or cover) (see Feoli et al. in this book, to understand the meaning of these terms and concepts).

6.2 Pedodiversity and biodiversity analysis: general comments

An intriguing feature appears when pedodiversity and biodiversity analysis are compared. Feoli et al. (in this book) reviews mathematical tools used in biodiversity analysis. In general ecologists have found several regularities in the assemblages of biological species in biocenoses, proposing their corresponding causation mechanisms in biological terms. However many of them also appear when the soil assemblages are analyzed (e.g., Ibáñez et al. 1990a, 2005a,b, Phillips 1999, 2001a,b, Phillips and Marion 2004, 2005, Toomanian et al. 2006, Toomanian and Esfandiarpour 2010). Therefore it seems that there are other underlying mechanisms that must be considered (see Section 6.7, and Phillips’ chapter in this book). In the following subsections, some of these mechanisms are explained.

Several studies show strong correlations between pedodiversity and biodiversity in several areas, different environments and distinct scales (e.g., Petersen et al. 2010, Ibáñez and Effland 2011). The same occurs between pedodiversity and landform diversity (e.g., Ibáñez et al. 1994, Toomanian et al. 2006, Toomanian and Esfandiarpour 2010). Furthermore, relations among biodiversity of different taxa, lithological diversity and climate diversity have also been demonstrated (e.g., Ibáñez and Effland 2011).

Phillips and Marion (2004, 2005) and Scharenbroch and Bockheim (2007) demonstrated a clear relationship between niches into forest and pedodiversity via biomechanical effects of tree fall using classical and numerical taxonomies.

Willis curve

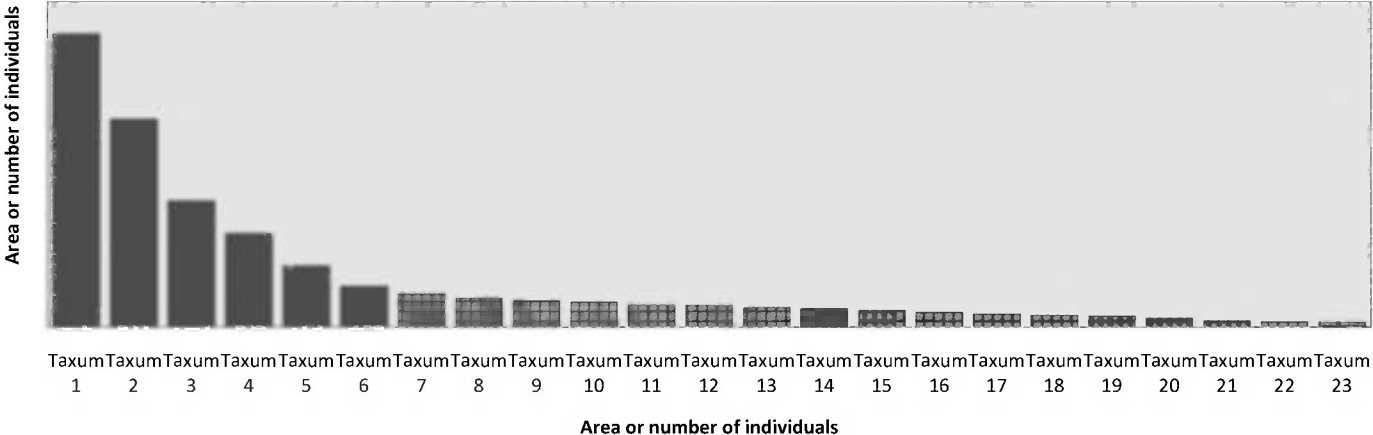


Figure. 1.4 Hypothetical hollow curve: plot of ranked-abundance list (area in km²).

6.3 Abundance distribution models

The distribution of the number of species within a community has long been regarded of great importance in the study of community structure. Although there are numerous models available in the literature, the following are most common: (i) geometric series; (ii) logarithmic series; (iii) log-normal distribution; and (iv) the “broken stick” model (Magurran 1988). The behaviour of these distributions is usually analyzed using rank/abundance plot in which objects are placed in order according to their decreasing degree of abundance, as is shown in the Fig. 1.4. There is a sequential order of distributions starting with the geometric series which is the least equitable (a few objects are dominant while the rest are very rare), continuing with the logarithmic series and the log-normal distribution and ending with the broken stick model (the most equitable, in which the taxa abundance within an assemblage is similar to a stick broken randomly and simultaneously into S units).

Biodiversity research shows that species in undisturbed ecosystems usually fit well to a log-normal distribution (e.g., Magurran 1998 and references therein), whereas under perturbation conditions logarithmic or geometric distributions are detected (see Tokeshi 1993 and references therein). Ibáñez et al. (1995, 2005b), Guo et al. (2003), Saldaña and Ibáñez (2004) and Scharenbroch and Bockheim (2007) show that the same regularities appear in soil assemblages.

6.4 Diversity-area relationships

Species-area relationships have been intensively studied in biodiversity studies, being also the theoretical core of conservation biology (e.g., Huston 1994, Rosenzweig 1995). The species-area relationships most often conform to a power law (see Chapter of Feoli et al. in this book). Thus, the logarithm of the number of taxa is proportional to the logarithm of the area. In a similar way Ibáñez and De-Alba (2000), Ibáñez et al. (2005a,b), Ibáñez and Effland (2011), Phillips (2001), Phillips and Marion (2004, 2005, 2007), Guo et al. (2003), Toomanian et al. (2006) as well as Toomanian and Esfandiarpour (2010) show that pedodiversity-area relationships are also fit by power laws, independently of the type of classification and scale employed (Fig. 1.5).

It has been demonstrated that in islands of different sizes in a given archipelago or analogous habitat of patchy spatial occurrence (e.g., lake-forest fragments and mountain-top niches), the exponent value of the power law function is often around 0.25 (e.g., MacArthur and Wilson 1967). It is noticeable that the same occurs when the number of pedotaxa is analyzed in islands of archipelagos (Ibáñez et al. 2005a, Ibáñez and Effland 2011).

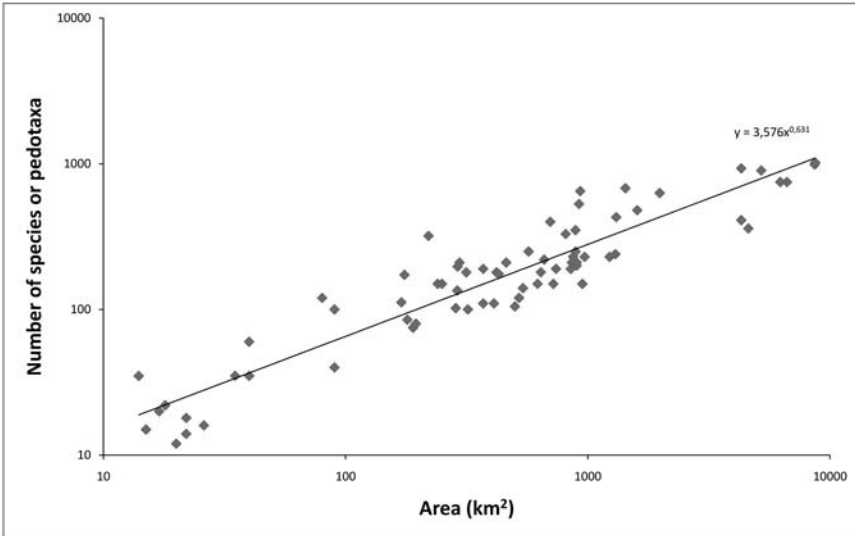
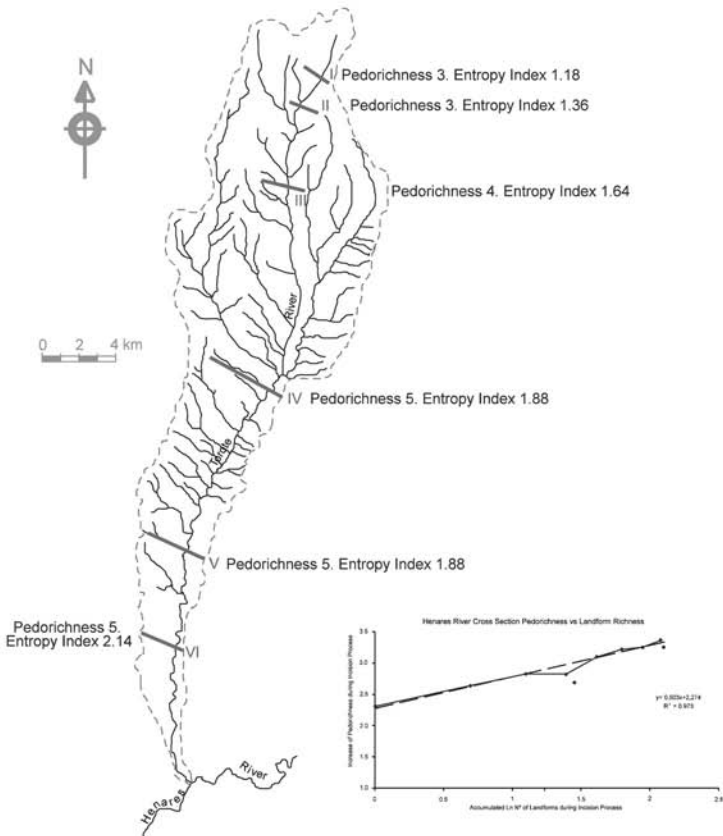


Figure 1.5 Richness-area relationships fit to a power laws (hypothetical example).

6.5 Diversity-time relationships

The increase of species diversity with time in natural undisturbed areas has been amply demonstrated in ecological literature (e.g., Rosenzweig 1995 and references therein). Saldaña and Ibáñez (2004) as well as Toomanian and Esfandiarpour (2010) detected pedodiversity-area relationships in chronosequences (fluvial terraces) that conform to power laws (but see also Phillips 2001b). Likewise Ibáñez and Effland (2011) demonstrate an increase of pedodiversity on time from the younger to the older islands at USDA Soil taxonomy order and suborder levels in the Hawaiian archipelago. However, this topic has been studied mainly by Jonathan Phillips and co-workers (e.g., Phillips 1999, 2001b) in the frame of divergent pedogenesis (see Sections 6.7, 6.8 and Phillips' chapter in this book). In addition, the studies of Phillips and Marion (2004, 2005) and Scharenbroch and Bockheim (2007) show how the architecture of forest soils increases its diversity in their interaction with the tree life-cycle with time (including tree falls). Furthermore Ibáñez et al. (1990a, 1994) and the De-Alba et al. (1993) show how the temporal process of fluvial incision and the hierarchization of drainage basins generate an increase of soil diversity (Fig. 1.6). Therefore as it occurs in the case of diversity-area relationships, the number of soils can and often does increase with time as ecological literature predict it for biological species (but see also Phillips 2001b).

PEDOLOGICAL SPATIAL VARIATION ANALYSIS: Torote River, a tributary stream of the Henares River



LITHOLOGICAL AND GEOMORPHOLOGICAL CHRONOLOGICAL VARIATION ANALYSIS: Henares River cross-sections

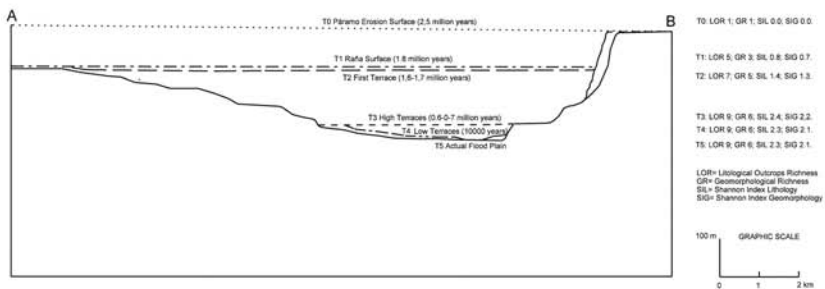


Figure 1.6 (A) Pedorichness and Entropy index variation along a tributary drainage basin of the Henares River. (B) Graph illustrating an accumulative increase of pedorichness from water interfluvial to the flood plain conforming fluvial incision was creating new landforms (current landscape). (C) Schematic paleoreconstruction of a Henares River cross-section (central Spain): Repercussion on pedodiversity, landform diversity and lithodiversity (raw data from Ibáñez et al. 1994).

6.6 Diversity and nested systems

A common pattern of species and soil types (pedotaxa) linked to species–area relationships is called “nested subsets” (Patterson and Atmar 1986). This pattern arises in view of taxa that appear on smaller assemblages (e.g., islands or drainage basins) also are present in other larger ones of the same type, but the opposite is not true; thus larger assemblages contain additional idiosyncratic pedotaxa that do not appear in the small ones (Fig. 1.7). The nested subset pattern arises because taxa differ in their distributions across space. Some taxa appear to be associated exclusively with idiosyncratic habitats. In pedological terms it is reasonably understood as a result of a variety of soil forming factors. Larger islands, on average, have more species and soil types than small ones because there is more variety in soil forming factors, such as relief, microclimates and mature depositional landforms (e.g., Ibáñez et al. 2005b and references therein). This fact is the result of two driving forces: (i) there is a positive correlation (a power law) between larger islands (or drainage basins) and relief that increases the environmental variety; (ii) larger islands and drainage basins have landforms that do not appear in the smaller ones (Ibáñez et al. 2005b). In pedological terms it is possible to predict the probability of that some specific soil types appear according to the size of the above mentioned land units increase (e.g., Ibáñez et al. 2005b and references therein). Nestedness patterns and pedodiversity loss induced by urban sprawl is explained by Zhang (Chapter 9).

Another pattern detected by ecologists is termed “taxa-range size distributions”. According to it, the abundance (or coverage) of a taxon and the extent of its spatial distribution are correlated. Widespread species are much more abundant than species of restricted occurrence in a territory. This pattern also conform to a power law, being corroborated to living organisms and soil types (e.g., Ibáñez et al. 2005b and references therein).

6.7 Diversity and complexity sciences

Studies of the relations between soil (Ibáñez et al. 1990a,b, Ibáñez and García-Álvarez 1991), pedodiversity (Ibáñez et al. 1990a), geomorphological systems and nonlinear systems (Phillips et al. 1992a,b) date back more than 20 yr. These authors analyzed pedological and geomorphological patterns as the results of their nonlinear nature. Phillips and co-workers began the study of the non-linearity of geomorphic systems in depth (Phillips 1992a,b) publishing many papers over the years. Ibáñez and colleagues (1990a,b, Ibáñez and Ibáñez and García-Álvarez 1991) begun to work on the same task with respect to soil genesis using non-equilibrium thermodynamics and catastrophe theory perspectives (Ibáñez et al.

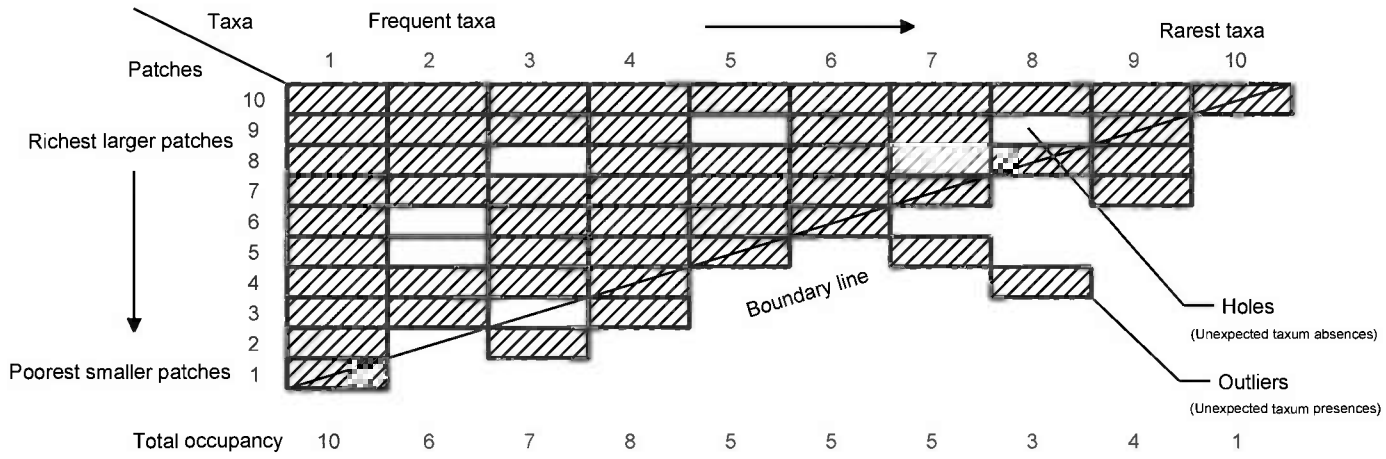


Figure 1.7 Perfect and realistic nested matrices (modified of Ibáñez et al. 2005b).

1990a,b). Finally Ibáñez et al. (1990a, 1994) discussed relations between pedodiversity, drainage basin structure and fluvial dissection, showing that as landscape dissection proceeds, soil diversity increases (Fig. 1.6 and Fig. 1.8). A few years later Phillips and co-workers began to analyze the relations between pedodiversity and landforms as nonlinear systems (e.g., Phillips 1999, 2001, Phillips and Marion 2004, 2005, etc.). Meanwhile Ibáñez and co-workers initiated studies for detecting similarities and differences between the patterns of biodiversity and pedodiversity (e.g., Ibáñez et al. 1994, 1995, 2005b, but see also Phillips 1999).

Non-linear and some complex systems may be characterized by sensitivity to initial conditions and local disturbances, and divergent evolution (see Phillips' chapter in this book). Many also show some scale invariant properties (fractals or multifractals as shown by San-José and Caniego' chapter in this book), as well as scale-dependent (emergent properties) ones. In the above mentioned publications, among others the authors explain that the obtained pedodiversity results could be considered as a "fingerprint" of the above-mentioned nonlinearity.

6.8 Diversity and pedogenetic theories

The dynamics of complex systems show a different perspective of pedogenesis with respect to classical ones. Traditional pedological theory holds that with similar initial conditions and environmental history soil genesis should follow a convergent developmental pathway to a certain soil climax (e.g., Jenny 1941). Thus, according to Jenny's theory mature soil landscapes develop with time, and this convergence results in a decrease in pedodiversity (but see Johnson and Watson-Stegner 1987 for an alternative point of view). In contrast, the nonlinear dynamics approach demonstrates that both divergent and convergent pathways could occur under the above-mentioned conditions (e.g., Phillips 1998, 1999, but see also Ibáñez et al. 1991) and could generate an increase of soil diversity.

Likewise, while classical pedology focused its attention on the evolution of soil types, nonlinear dynamics, complexity sciences and pedodiversity tools additionally permitted the mathematical analysis of the genesis of soils. Thus, using pedorichness-area-relations, Phillips and co-workers (e.g., Phillips 2001, Phillips and Marion 2004, 2005) show that intrinsic factors (instability of soil systems as a driving force of divergent pedogenesis) at least are as important as extrinsic ones in understanding pedogenesis along the space and time axes, with an increase in pedodiversity on a given landscape, particularly for fine-resolution scales (e.g., Phillips 2001a,b, Phillips and Marion 2004, 2005, among others, and Phillips' chapter in this book). This fact could be also understood using species-time relationships as shown in Section 6.5.

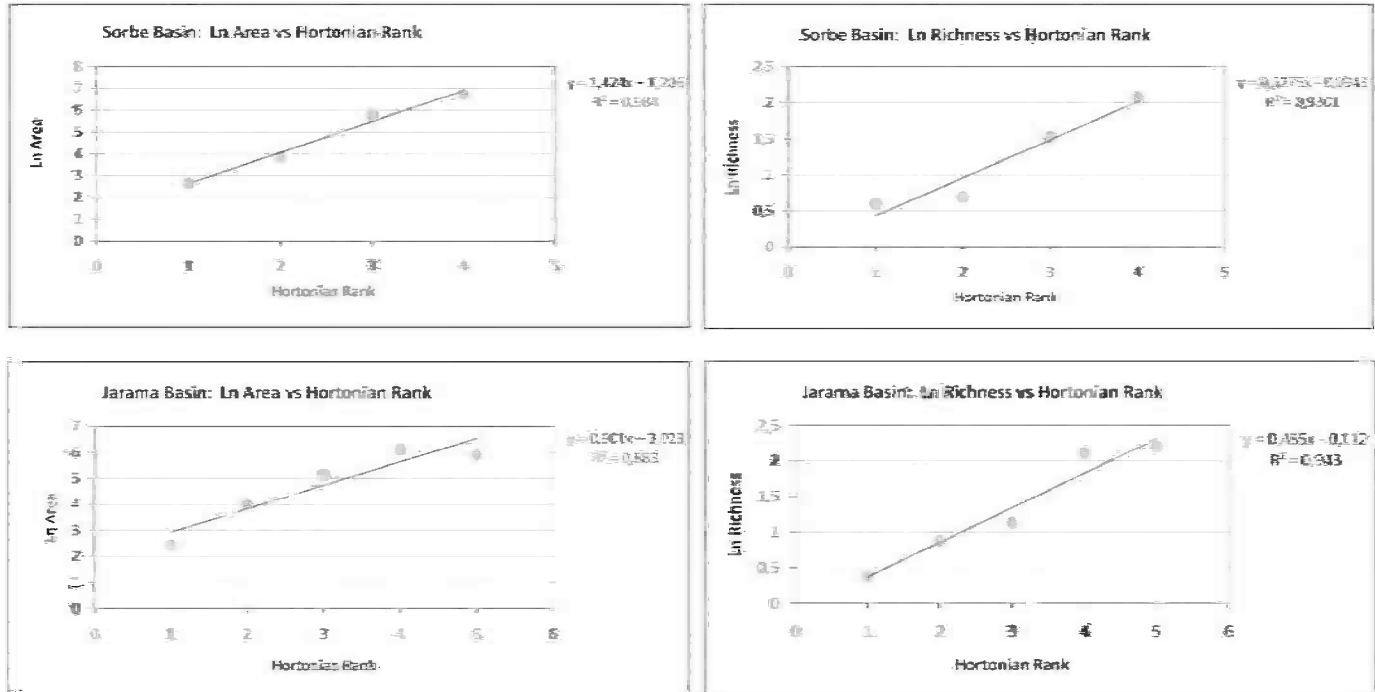


Figure 1.8 Pedodiversity-area relationships of two drainage basins in the “Macizo de Ayllón” Mountain Range (Central Spain) at fine scales (1:20.000) (Raw data from Ibáñez et al. 1990a). The hortonian rank or stream order is positively correlated with the mean basin area. Likewise basin area-pedodiversity relationship fits to a power law. Therefore the hortonian rank-pedodiversity relationship also conform to a power law (Horton 1945, Strahler 1952, Ibáñez et al. 1990a).

6.9 Diversity, fractals and multifractals

Ibáñez et al. (2005a,b) and Ibáñez and Effland (2011), using pedorichness-area relationships, conjecture the fractal distribution of soil types across geographical spaces. Likewise Caniego et al. (2006, 2007) made use of multifractal tools to analyze the scale invariance properties of the pedosphere structure at global scales. These studies show the scale-invariant properties of the spatial distribution of soil types or pedotaxa. Intriguing relationships between several diversity indices and multifractal structures are explained by San-José and Caniego in Chapter 4 of this book.

6.10 Diversity, biogeography and pedogeography

The theory of island biogeography (MacArthur and Wilson 1967) is the keystone of theoretical biogeography as well as an indispensable tool in conservation biology (e.g., Huston 1994, Rosenzweig 1995). This theory predicts, using biological assumptions, that in islands species-area curves conform to a power law which has an exponent of 0.25 in islands. Ibáñez et al. (2005a) found that pedorichness-area relationships have the same statistical distribution and the same exponent and are the product of the nonlinearity of the spatial distribution of soils. Therefore, these results refute the well-established idea in ecology that the MacArthur and Wilson theory rests only on biological assumptions, such as species migration and extinctions, distance from the islands to a mainland as source of gene pools, etc. Ibáñez and Effland (2011) proposed a theory of island pedogeography in which the driving forces of soil assemblages and biological communities on islands are plate tectonics (and their repercussions on lithology, and landforms) as well as the latitude. Thus these authors propose a unifying theory of biodiversity and pedodiversity, where pedodiversity and biodiversity values are positively correlated.

6.11 Diversity and landscape evolution

Arnett and Conacher (1973) and Conacher and Dalrymple (1977) showed that after tectonic surface uplifting fluvial erosion and development of river networks produced a progressive increase in the heterogeneity or number of pedogeomorphological units. Ibáñez et al. (1990a) analyzed a similar case in the frame of pedodiversity and the theory of complex systems, arriving at similar conclusions. Thus the latter authors making use of the Shannon entropy index (see the chapter of Feoli et al. in this book) detect that the number of soil associations present in river basins increase with the complexity of the drainage networks, from 1.93 to 4.20 bits. Ibáñez et

al. (1990a) interpret the network structuring, its relief repercussions and the increase of soil diversity as different aspects of the same process: the self-organization of complex systems with time. In addition these authors conjectured that the same should occur with the plant communities that evolved on these landforms and soilscares. Hupp (1990) reached a similar conclusion: a positive relation exists between the progressive development of the river networks, the number of geomorphological units, and richness of flora that appear on the landforms.

Ibáñez et al. (1994) carried out a paleo-reconstruction of the evolution of a channel cross section during the last 2.5 m.y (Fig. 1.6). These authors showed how the network incision induced an increase in the richness and diversity of lithological, geomorphological, pedological and phytocenotic units in the study area. Thus, the incision process increases the number of geomorphological units around 75 percent and pedorichness in 51.5 percent with respect to old landform interfluves. Furthermore, this process is parallel with an expansion or dilatation of the geographical space. In other words, the length of the channel cross-section increased as geological erosion increased the roughness of the landscape (estimating its fractal dimensions).

Meanwhile Phillips (1992a and b) analyzed and corroborated the non-linear dynamics of geomorphological systems. There exist multiple lines of evidence to state that soilscares, landforms and biocenoses are coupled non-linear systems and thus biodiversity, pedodiversity and landform diversity trends follow the same patterns (e.g., Ibáñez et al. 1990a, Phillips 1999, Ibáñez and Effland 2011). The main difference between the approach of Phillips and co-workers and that of Ibáñez and co-workers is mainly the scale of their respective studies. The studies of Phillips team were conducted mainly at detailed scales (but see also Sadaña and Ibáñez 2004, 2007), so that they show the importance of divergent pedogenesis on soils systems that are caused by minor differences in the initial conditions in short distances. In contrast the latter authors considered broader scales and larger territories and showed how extrinsic factors affect soilscares. These multiple perspectives seem to be complementary approaches.

7. Pedodiversity and Preservation of the Pedosphere

7.1 Pedodiversity at risk

The human impacts on land surfaces, soilscares and ecosystems have increased dramatically over time. As with biodiversity, pedodiversity is also in danger. Anthropic perturbations are putting global pedodiversity at risk (e.g., Dazzi and Monteleone 1999, Lo Papa et al. 2011, Amundson et al. 2003, Zhang et al. 2007, etc.). Amundson et al. (2003) documented

several soil series in risk of extinction in EE.UU. Likewise Lo Pappa et al. (2011) showed the loss of pedodiversity in Sicily over a 53-yr period, as well as dramatic predictive scenarios in 2050 using neural Markov chains and cellular automata (see Lo Papa and Dazzi in this book).

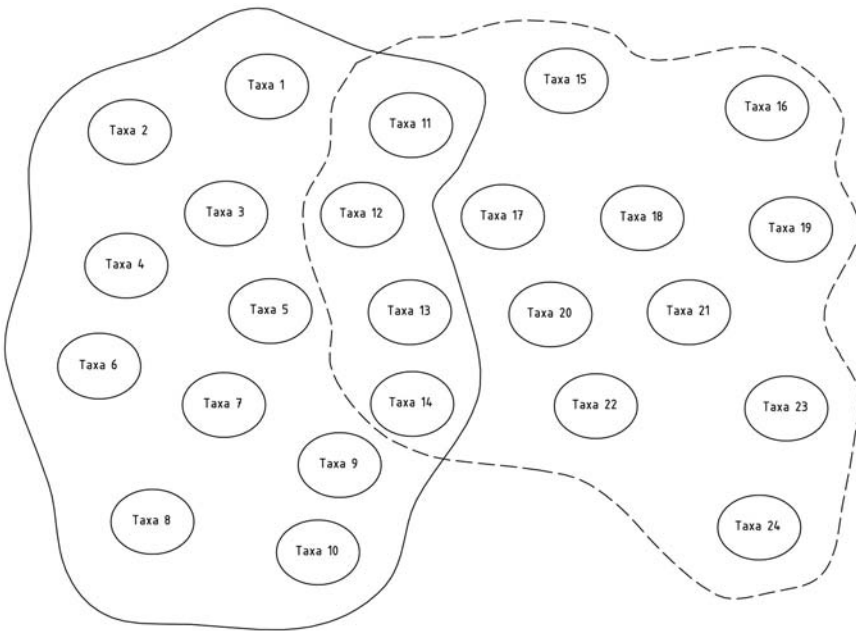
Zhang et al. (2007) demonstrated the loss of pedodiversity as a result of urban sprawl and infrastructures in densely populated industrial areas. In contrast Dazzi et al. (2009) and Lo Pappa et al. (2011) studied perturbations caused by industrial farming practices in rural territories. The latter authors also detect the genesis of new soil types (manmade soils) at the expense of natural ones. The current decrease in natural pedodiversity alerts us to the devastating effects of human beings in modifying natural soilscapes. Finally, it has been reported that contaminated materials flooding from metalliferous mines cause a significant impact on functional and taxonomic pedodiversity in respect to previous and surrounding natural areas (Vacca et al. 2012).

7.2 Pedodiversity and soil preservation

Because the crucial role of soils for human survival, pedodiversity preservation merits special attention, perhaps more than other natural resources. However, to date national and local policies do not pay much attention to the conservation of this vital resource for the biosphere. Pedodiversity is part of our natural (geodiversity, soil biodiversity) as well as cultural heritages as is showed in Fig. 1.1 (Ibáñez et al. 2008), being a block of memory of the past landscapes and environments (e.g., Bockheim and Haus in this book). Likewise soil biodiversity has been poorly studied, though many experts recognize that it is likely greater than that aboveground (e.g., Usher 2005). According to Ibáñez et al. (2008) this fact is intriguing because: (i) a major part of the biodiversity of terrestrial ecosystems is housed in soil, either totally (e.g., microbial and fauna soil communities) or partially (e.g., underground biomass of plant communities, habitat of reptilians and some small mammals); (ii) some soils and/or soilscapes are essential to the conservation of certain biological species (edaphic endemism) and plant communities: (e.g., Bockheim and Haus' chapter in this book), because if the former disappear, then the latter may disappear; (iii) the soil system is vital in ecosystem and food web dynamics; (iv) there is a need to determine benchmark soils whose "soil functions" remain as undisturbed as possible as sites of reference for studies on soil quality monitoring programs; (v) soil endemism must be preserved as part of the pedological heritage (e.g., Amundson et al. 2003, Bockheim 2005) and (v) as we stated in the previous section many pedotaxa are at risk of extinction as a consequence of urban sprawl, mineral extraction, agriculture and other human-induced processes. Therefore large parts of the pedosphere consist of "domesticated soils" (Amundson

et al. 2003). The study and importance of the endemic soils are analyzed in depth by Bockheim and Haus in this book.

Ibáñez et al. (2008) analyzed which soil resources must be preserved, as well as the potential strategies for designing soil reserve networks. These authors promote the advantages of complementary methods similar to strategies used in conservation biology (e.g., hot spots and rarity areas). Complementary methods guarantee the conservation of all soil types and assemblages using the minimum area as possible in a given territory. This search sets complementary land units, which contain all the pedotaxa in a given geographical space. Currently, conservation biology applies several useful algorithms for reserve selection. An example of the rationale underlying the complementary areas framework in a pedological context (Fig. 1.9) could be the following: if one area has a spatial soil assemblage consisting of Cryosols, Histosols and Gleysols, and the other area has Histosols, Gleysols and Podzols, then the second one complements the first one with the Podzols taxon (Ibáñez et al. 2003). While hotspots of



PRINCIPLE OF COMPLEMENTARITY

Figure 1.9 Diagram illustrating the conceptual scheme about how complementary methods work.

richness and rarity would fail to represent “all undisturbed taxa” at least once, complementary areas permit representation of all taxa at least twice in a given percentage of the study area. Ibáñez et al. (2003) provided a pedological test of complementary methods in the Aegean Archipelago.

8. Concluding Remarks

As has been demonstrated in previous sections, pedodiversity is a growing industry, being accepted as a valuable pedometric tool McBratney et al. (2000). Several interesting spatial and temporal patterns have been detected using the pedodiversity analysis that illustrate the importance of natural-resource inventories and preservation. Likewise the mathematical tools used by experts for the study of the soil diversity distribution are also applied in order to quantify soil maps as well as contribute to theoretical pedology. For example, a theory of island pedogeography has been proposed. Likewise the old conception of convergent pedogenesis has been refuted or expanded by other authors, who show that divergent pedogenesis take place in many instances. However current trends in the literature offer reasons to be concerned.

Biodiversity analysis has a long tradition, and several authors as Hurlbert (1971), Peters (1991) and Ricotta (2005) have analyzed the progress, advances and controversies on the state of the art at different times in the last several decades. Perhaps it is time to carry on a similar intellectual exercise in pedodiversity analysis. For example, Ricotta (2005) stated that *“Biological diversity would apparently seem the most intuitive and easily studied of all the ecological concepts. However, in practice biodiversity has suffered from great number of definitions that vary with the specific needs of the different researchers, thus making it extremely confusing as an ecological concept. (...) there exists a substantial ambiguity among ecologists as far as biodiversity conceptualization and evaluation is concerned”*. This statement is equally valid in the framework of pedodiversity analysis. It is paradoxical that along with the growth of the literature on soil diversity, pedologists are repeating the same mistakes as ecologists. The proliferation of concepts, indices, and other mathematical tools are hindering rather than advancing the progress of this line of research, in view that few studies made use of the same protocols. Under this situation the comparison of results in order to detect pedodiversity patterns is hampered. Because of this, some ecologists claim that biological diversity or biodiversity is a “nonconcept” (Hurlbert 1971). Along these lines the editors of this book have tried to avoid this trend. An example follows.

The Shannon diversity index has been harshly critiqued in ecological and pedological literature (e.g., McBratney and Minasny 2007, Petersen et

al. 2010, Toomanian and Esfandiarpour 2010). However, it is also the most widely used and enduring index in biodiversity analysis (Magurran 2004, Ricotta 2005). As with all diversity indices, the Shannon index has several constraints, many of which also apply to hundreds of other diversity indices that have been proposed over the past decades. When other authors propose a new index, they usually compare it with the Shannon algorithm rather than these other indices. This proliferation of proposals has not led to better estimates of diversity; rather, it has led to methodological chaos, as well as sour and endless polemics (e.g., Hurlbert 1971, Ricotta 2005). To date there is no consensus on indices to replace the Shannon index. From a strictly mathematical point of view, the Shannon index is a valid algorithm for the quantification of diversity (Martin and Rey 2000). Hurlbert (1971), Peters (1991), Ricotta (2005), and Ibáñez et al. (1990a) all strongly criticized the proliferation of indices in biodiversity studies. Thus, as pedologists we are not learned from the mistakes incurred by the biodiversity experts. The search for new algorithms is a legitimate task, however it should not compete nor replace the fundamental detection of patterns.

In this chapter we have tried through a literature review to show efforts made to identify clear and consistent pedodiversity patterns. The search of regularities or patterns should be the main focus of pedodiversity and biodiversity analysis. This should be accompanied by an examination for scientifically sound and undisputed diversity indices. Meanwhile there are some intriguing questions that have not been solved yet: (i) why are pedodiversity results similar to the biodiversity ones?; (ii) are these similarities a consequence of the non-linear dynamic nature of these earth-surface systems?; (iii) how does pedodiversity compare to the diversity of other natural resources such as geomorphological units (e.g., Arnett and Conacher 1973, Conacher and Dalrymple 1977, Phillips 1999, 2001a,b, Phillips and Marion 2007, Toomanian and Esfandiarpour 2010) and lithological units as discussed by Williamson (1981) and Ibáñez et al. (1994)? In our opinion these are the main challenges of pedodiversity analysis (Fig. 1.10).

On the contrary, if methodological tools are the main focus of our attention, pedologists could claim, paraphrasing Ricotta (2005) that "*biodiversity (pedodiversity) may be defined simply as a set of multivariate summary statistics for quantifying different characteristics of community structure (soil assemblage structures)*".

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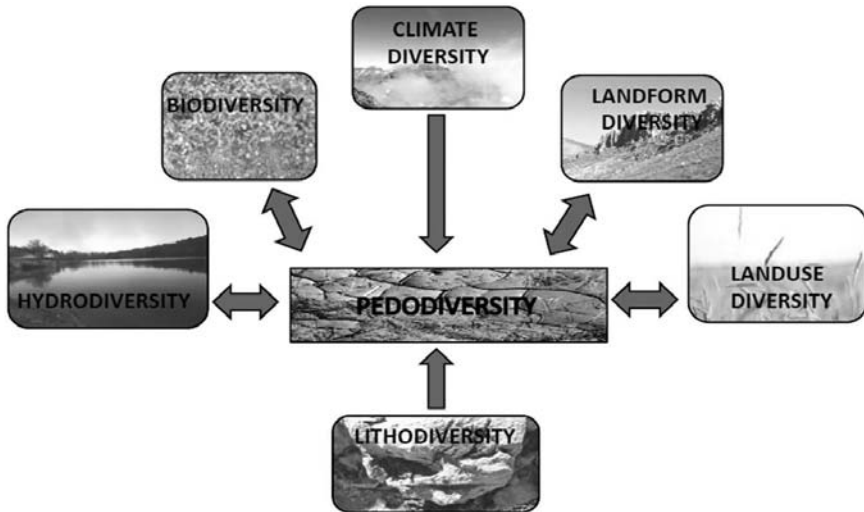


Figure 1.10 Diagram illustrating the relations between pedodiversity, biodiversity, landform diversity, lithodiversity, climate diversity, hydrodiversity and landuse diversity (soil forming factors).

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CHAPTER 2

Measuring Diversity of Environmental Systems

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1. Introduction

Diversity is a controversial concept and its common use in different scientific disciplines (from chemical-physical ones to ecological and socio-economical ones) and in our everyday language, has led to many discussions about its meaning especially in ecology (e.g., Hurlbert 1971, Feoli et al. 1988, Ricotta 2005, Tuomisto 2010a and b, Jurasinski and Koch 2011, Moreno and Rodríguez 2011, Tuomisto 2011). The great attention that the concept of diversity (biodiversity) has in ecology is due to the fact that it is considered an important characteristic and a useful parameter of the biotic component of habitats, landscapes, biomes and biosphere to understand and infer ecosystem stability, the assembly rules of the plant and animal communities and their changes in space and time. The importance given to the concept in ecology is testified by the large number of papers and books dealing with the concepts and formulas for measuring biodiversity under different perspectives (e.g., Whittaker 1972, May 1975, Pielou 1975, Wilson 1992, Magurran 1988, 2004, Ganis 1991, Huston 1994, Levin 2001, Orlóci et al. 2002, Mason et al. 2005, Jost 2006, Würtz and Annala 2008, Hillebrand and Matthiessen 2009, Pavoine and Bonsall 2011, etc.).

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As Orlóci et al. (2002) point out that biodiversity is a unifying concept in ecology being “simultaneously a cause and an effect, in the functioning of the global ecosystem”, it can be said that the concept of diversity could have an important role of “unifying” science being the study of the “diversity” of nature in all its aspects. We do not intend to enter into philosophical discussions or those of any particular discipline, but we would like to only review and comment on some ways of measuring diversity of environmental units, sampled from the environmental systems, under the perspective of the concept of similarity-dissimilarity (Feoli and Orlóci 2011). The environmental units may be described by sets of events or objects (O) of any type, material or immaterial (species or other taxa and/or functional groups, land cover types, pedotypes, energy types, social types, ethnic groups, types of job, types of books, ideas, political parties, etc.). The responsibility of defining the framework in which the measures can be done is left to the researchers. It is clear that to measure something does not mean to know what we are measuring!

We also do not want to be involved in the discussion of the terminology concerning alpha, beta, gamma and delta diversity of Whittaker (Whittaker 1972, Magurran 1988, 2004, Jost 2007, Tuomisto 2011) that are particularly active in ecology (e.g., Podani and Schmera 2011) and that would also be relevant in other disciplines, for clarity we prefer to call diversity only the “diversity” of an entity corresponding to a sampling unit, namely what is called alpha diversity or gamma diversity (alpha diversity for a primitive sampling unit, gamma diversity for a sampling unit obtained by grouping a certain number of primitive sampling units), leaving the terms beta diversity and delta diversity to the domain of measuring similarity-dissimilarity between sampling units, that is well explored in numerical taxonomy and data analysis in general (e.g., Sneath and Sokal 1973, Orlóci 1978, Legendre and Legendre 1998, Podani 2000, Feoli and Orlóci 2011) and in artificial intelligence (see the web). In other words we want to stress that by the term diversity we consider only: “how many different things (objects, events) are in a given collection (a sampling unit) and how much such a collection is heterogeneous in terms of such objects” and not “how much diverse two different sampling units are”. Accordingly we suggest the following general definition of diversity: “Diversity is a property of a sampling unit containing a set of events or objects (O) of any kind, material or immaterial, real or imaginary, of which $C(O)$ is a partition, measurable by numbers that combines the number of classes (S) of $C(O)$ and the average dissimilarity between the S classes”. In other words diversity is a function of two variables, the number of classes (S), namely the richness of a partition $C(O)$, where each class includes events considered similar enough to belong to the same class, and the average dissimilarity (Δ) between the classes or the heterogeneity of $C(O)$. It follows that given two units described by sets

of events (O1) and (O2) with the same number of classes S , the diversity of (O1) will be higher than the diversity of (O2) if Δ of (O1) will be higher than the Δ of (O2). We also can expect that a unit described by a set of events (O1), less numerous than that of unit described by (O2), could have higher diversity than that described by (O2) if the Δ of (O1) is much more higher than the Δ of (O2). It follows that the measure of diversity finds its meaning only in well defined contexts, where the criteria to define the classes and to measure the dissimilarity between them are clearly defined. On the basis of the above definition, we want to simplify the matter by defining two kinds of diversity: crispy diversity and fuzzy diversity. Crispy diversity corresponds to the situation in which the S classes of $C(O)$ have the within similarity equal to 1 and the between similarity equal to 0. In this case the diversity of (O) depends only on the number S of the classes of $C(O)$ and on the proportionality between them (evenness). Fuzzy diversity is defined when we decide on considering the within and between similarity-dissimilarity of classes with a resemblance function ranging between 0 and 1 calculated on the basis of a selected set of characters that we want to use for describing the objects. We will show that the unifying notation of diversity in terms of richness and evenness as suggested by Hill (1973) is useful for a unified notation concerning richness and similarity-dissimilarity in line with Rao (1982), Ricotta and Szeidl (2006, 2009) and Leinster and Cobbold (2012). We also present some alternative formulas of parametric diversity and non parametric diversity and discuss the importance of hierarchical analysis of diversity and the use of similarity-dissimilarity functions and the dendrograms for detecting and measuring diversity of an environmental sampling unit.

2. Diversity-area Relationships of Environmental Systems

Diversity in environmental science is by definition a spatial dependent phenomenon. When we are studying an environmental system, we always consider implicitly the area or volume occupied by it (the spatial dimension). In general we can expect that the number of events that can be examined in a sampling unit of the system would increase as the area or volume of the unit increases. In environmental sciences there are many studies related to species–area relationships (SAR) under different perspectives and different spatial and organizational scales (e.g., Connor and McCoy 1979, He and Legendre 1996, Wright 1983, Scheineir et al. 2000, Drakare et al. 2006, Würtz and Annala 2008, Harte et al. 2009) while species-volume relationships are still little explored being limited to aquatic ecosystems or tropical rainforests (e.g., Angermeier and Schlosser 1989, Davis and Sutton 2011). Richness-area relationships have also been studied extensively in pedodiversity (Ibáñez et al. 1990, Ibáñez and

De-Alba 2000 and Ibáñez et al. 2009) however the diversity-area relationship is still neglected. In landscape ecology it is usual to calculate the diversity of the landscape of a given territory considering the number and the extent of the areas of different land types (e.g., Turner 1990, Hoover and Parker 1991, Farina 2000, Turner and Gardner 1991, Willems et al. 2000), however it appears that this kind of diversity has never been considered in function of the extent of the area (land cover diversity-area curves). In any case curves diversity-area can be easily calculated for many purposes and in different disciplines (e.g., analysis of industrial areas, urban areas etc.) once we assign abundance values to the events (e.g., land cover) and to their classes. They may be fitted by several functions, the most used one in case of richness-area curves is $S = A^z$, where A is the area sampled and z an empirical constant that for species-area curves usually ranges between 0.1 and 0.4 (May 1975, Drakare et al. 2006). This range is also accepted for pedotypes-area relationships (Ibáñez et al. 1990, Ibáñez and De-Alba 2000 and Ibáñez et al. 2009).

3. The Abundance and the Weight of the Classes

While the richness of (O) may be calculated only in one way, i.e., by counting the classes of $C(O)$, the evenness depends on the values we give to the abundance of the classes and on the formula used to measure diversity. Let us consider first the way to assign abundance to the classes.

This may come out from measurements made by objective methods based on physical instruments or by subjective judgements as in the case of species cover in plant communities. In view of this subjective judgement we prefer to consider the abundance as the relative weight given to the S classes in terms of a quantity Q that is measured for each class. Q can be the frequency of the events within the classes, values of biomass, the energy the classes are consuming, the area or the volume occupied by the classes, etc. It is clear that the assignment of weight to events and to the S classes remains a responsibility of the researcher. In any case when the quantity Q is established weight w_i is assigned to each class of $C(O)$ as $w_i = Q_i / \sum Q_i$ where Q_i is the quantity for the i th class, it follows that $\sum w_i = 1$. The value w_i may be considered as an estimation of the probability p_i to find something related to the i th class when we make a random sampling in the hypothetical "perfect" mixture of the events of (O). For example if we consider a mixture of individuals belonging to S classes of living organism, that may be species or other taxonomic or structural-functional groups, and if the weight of each class is given by the number of individuals in the class, the proportion of individuals p_i of a class is the probability to find an individual of that class when we make a random sampling of one individual out of the mixture. If the weight is given in grams or other units of biomass rather than by

individuals, then p_i is estimating the probability that a random sample of an amount of biomass equal to the unit of biomass will be “material” of the living organisms of the i th class. If the classes are the types of food that constitute the diet of an organism and if the weight is given in joules, the p_i will give the probability that the organism is consuming one joule of the i th class of food. If the classes are habitats in which an organism is living and the weight is the frequency or the abundance of the organism in the habitats then p_i will estimate the probability to find the organism in the i th habitat. If the classes are land types (e.g., land cover type, land use type, pedotypes, vegetation types etc.) of a given territory and the weights are given by the area measured in square meters, the p_i will estimate the probability that a square meter of land, randomly sampled in the territory would belong to the i th land type; if the weight is given by the number of pixels or cells (of equal size) of a grid of a land map then p_i is the probability to find a pixel or a cell of the i th land type in the map. And so forth. The most common graphical representation of the $C(O)$ are the rank/abundance histograms, called dominance/diversity curves, where the abscissa indicates the classes S disposed according to their rank of abundance, from the most abundant one to the least abundant, and the ordinate shows the corresponding abundance values (Wilson 1991) and histograms indicating the number of classes of $C(O)$ in correspondence of classes of abundance generally defined as octaves (Preston 1948, Magurran 1988). In different disciplines these curves are tested against models that would have specific meaning for each of them. Divergence from the models would suggest if there are specific laws that may describe the diversity pattern of a sampling unit or of a given set of sampling units. In ecology the most common models are the broken stick, the log-normal the log series, the geometric series, in other disciplines such as linguistics and economy the most used is the Pareto-Zipf-Mandelbrot model that seems to fit a great number of dominance/diversity curves also in other fields (Izsak 2006). The divergence is calculated with the chi-squared test or in terms of a $2T$ multiple of mutual information (Kullback 1959, Renyi 1961, Feoli et al. 1984, Orłóci et al. 2002) that approximates the chi-square distribution.

4. Parametric Measures of Diversity

Several functions have been proposed to measure species diversity and biodiversity (cf. Magurran 1988, 2004, Pavoine and Bonsall 2011) and can be classified into two broad categories, the parametric and the non parametric ones. The difference between the two categories is due to the fact that with a parametric function we can get directly different measures of diversity that are less sensitive to richness and more to dominance (the contrary of evenness) as the value of the parameter increases. These

functions may be applicable to any set of classes of events $C(O)$ for which the weight can be made relative to the total weight, i.e., $p_i = w_i = Q_i / \sum Q_i$, with $\sum p_i = 1$.

4.1 Crispy diversity measures

These measures correspond to the formulas of entropies (H) of Rényi (1961), Patil and Taillie (1976) and of Hill (1973) that are respectively known as alpha-diversity, beta-diversity and N-diversity.

The first is:

$$D(O) = H_\alpha = \frac{\ln \sum_i p_i^\alpha}{1 - \alpha} \quad 1)$$

The second is:

$$D(O) = H_\beta = \frac{(1 - \sum_i p_i^{\beta+1})}{\beta} \quad 2)$$

The third is:

$$D(O) = N_a = \left(\sum_i p_i^a \right)^{\frac{1}{1-a}} \quad 3)$$

The parameters α , β and a define an infinite number of possible “point measures” of diversity for the corresponding formulas, they are also called sensitivity parameters (Leinster and Cobbold 2012), because they make the value of diversity more sensitive to dominance and less to richness as far as their values are settled positive and far from 0 in case of α and a , and are settled bigger than (-1) in case of β . Considering the limits of the function 1) for $\alpha \rightarrow 0$ we obtain the $\ln S$ (i.e., the \ln of the richness) that is the maximal entropy, for $\alpha \rightarrow 1$ we obtain the well known formula of Shannon:

$$D(O) = H(O)_{\alpha_1} = -\sum_i p_i \ln p_i \quad 4)$$

for $\alpha \rightarrow 2$ we obtain the $-\ln$ of the well known Simpson index:

$$D(O) = H(O)_{\alpha_2} = -\ln S(E) = -\ln \sum_i p_i^2 \quad 5)$$

Considering the limits of the function 2) for $\beta \rightarrow -1$ we obtain the $S-1$, for $\beta \rightarrow 0$ we obtain the formula 4) (i.e., the Shannon index) and for $\beta \rightarrow 1$ we obtain the index known as diversity of Gini:

$$D(O) = H(O)_{\beta_1} = 1 - \sum_i p_i^2 \quad 6)$$

Considering the limits of the function 3) for $\alpha \rightarrow 0$ we get $N_0 = S$, for $\alpha \rightarrow 1$ we obtain:

$$D(O) = H(O)_{N_1} = e^{-\sum_i p_i \ln p_i} \tag{7}$$

for $\alpha \rightarrow 2$ we obtain:

$$D(O) = H(O)_{N_2} = \frac{1}{\sum_i p_i^2} \tag{8}$$

Of the three general functions only the third one reaches the maximum equal to S , that is the number of classes of the partition $C(O)$. This happens when there is equal proportion among the classes. This is seen as an advantage by Hill (1973) and Jost (2006, 2007, 2009) and is in line with our definition of diversity. Another important property of formula 3) is the fact that for $\alpha \rightarrow \infty$, $D(O)$ is equal to the number of classes of $C(O)$ if all of them would be weighed with Q_{imax}/Q , i.e., $D(O)$ would be Q/Q_{imax} , and in case of $\alpha \rightarrow -\infty$ $D(O)$ would be Q/Q_{imin} . In all the three general functions, diversity is decreasing in function of the average crispy similarity of the set of events (O) , in fact we can say that as the parameters α , β and γ increase respectively to values 2, 1 and 2, in the formulas 1), 2) and 3) the formulas 5), 6) and 8) gives values of $H(O)$ expressing more explicitly the quantity $\sum_i p_i^2$. It is easy to demonstrate that this value is the average similarity of a crisp similarity matrix in which the 1s represent similarity and the 0s dissimilarity. The elements of (O) can be individuals or units of measurements (grams, square meter, joules, etc.) and if $\sum Q_i$ is transformed in an integer number T representing the total number of units, these can be disposed into a matrix \mathbf{R}_{TxT} ($\mathbf{R} = \mathbf{1} - \mathbf{\Delta}$, where $\mathbf{1}$ represents a matrix $S \times S$ of 1s) of S blocks corresponding to S disjoint sub-matrices of \mathbf{R} . Since the S sub-matrices contain, according to the definition of crispy diversity, all values equal to 1, it is clear that the overall average similarity will be $(n_1^2 + n_2^2 + \dots + n_n^2)/T^2$, with n_i indicating the number of units for the i th class; if we consider $i = 1, \dots, S$, the average similarity value can be written as $\sum_i (n_i/T)^2$ and since $p_i = n_i/T$, it is clear that $\sum_i p_i^2$ is the average similarity of \mathbf{R} . It follows that the Gini index (formula 6) is the average crispy dissimilarity of the events in (O) . For a given number of S classes the minimum average similarity will be $R_{\text{min}} = S/S^2$ (i.e., $1/S$, because the similarity matrix \mathbf{R} is an identity matrix) consequently the maximum average dissimilarity will be $\Delta_{\text{max}} = 1 - 1/S$, while the maximum average similarity will tend to 1 when the quantity $((T-S+1)/T)$ is concentrated in only one class, leaving to the other classes just one unit of Q , i.e., $1/T$, so that $R_{\text{max}} = ((T-S+1)/T)^2 + (S-1)(1/T)^2$. It is clear that the situation of a maximum diversity of a set (O) of S objects is reached when the objects are completely dissimilar. The complement of the identity matrix is a matrix with 0s in the diagonal and 1s in the other cells.

If we consider $R(O) = \sum_i p_i^2$ as the average crispy similarity, the formulas 5) 6) and 8) can be written respectively as $-\ln R(O)$, $1-R(O)$ and $1/R(O)$.

It is easy to demonstrate that the eigenvalues λ_s of a crisp similarity matrix \mathbf{R} organized in S disjoint blocks are the square root of the number of the cells in the blocks, therefore the n_i is the eigenvalues λ_i of \mathbf{R} that corresponds to its i th block. Based on such property of the crisp similarity matrices \mathbf{R} , the p_i s of $C(O)$ may be interpreted as the proportions of the S eigenvalues of the matrix \mathbf{R} . Feoli et al. (2009) propose using the evenness of the positive eigenvalues of a similarity matrix as a measure of sharpness of a given classification. In terms of diversity the index proposed by Feoli et al. (2009) is nothing else that the relative diversity of a given $C(O)$ and the approach based on classification of a given set of entities belonging to a given system may be useful to measure the diversity of the system at different hierarchical levels of its description. This will be further dealt with in the paragraph on hierarchical diversity profiles. In summary it can be considered that the formula $\sum_i p_i^2$ expresses the average crispy similarity of $C(O)$ and we can use the terms similarity or homogeneity as synonymous of dominance.

4.2 Fuzzy diversity measures

The classical way to represent and measure fuzzy diversity was introduced by Rao (1982, 2010) and used in different contexts including pedodiversity studies (e.g., Clarke and Warwick 1998, Botta-Dukat 2005, Ricotta and Szeidl 2006, McBratney and Miasny 2007) notwithstanding he does not call his measure as a fuzzy measure. Given a set of S classes of (O) , Rao (1982) suggests to calculate the diversity of the set by the following equation:

$$D(O) = \sum_i \sum_j \delta_{ij} p_i p_j \quad 9)$$

or in matrix form:

$$D(O) = \mathbf{P}' \mathbf{\Delta} \mathbf{P}$$

in which \mathbf{P} is the vector of the n proportions of weights (rows of the vector) of the S classes of events $C(O)$ with $\sum_i p_i = 1$, and $\mathbf{\Delta}$ is a symmetric dissimilarity matrix ($n \times n$) in which the entries δ_{ij} are ranging between 0 and 1. It is easy to prove that if there is no similarity between the classes, i.e., $\delta_{ii} = 0$ and $\delta_{ij} = 1$, $D(O)$ is a crispy diversity equal to the index of diversity of Gini (formula 6).

If instead of considering a matrix of dissimilarity we consider a matrix of similarity \mathbf{R} , the equation (9) becomes:

$$R(O) = \mathbf{P}'\mathbf{R}\mathbf{P} \tag{10}$$

and if $r_{ii} = 1$ and $r_{ij} = 0$

$$R(O) = \sum_i p_i^2 \tag{11}$$

i.e., the index of Simpson, representing a measure of similarity or dominance rather than a measure of diversity, $R(O)$ is equal to 1 if there is only one class in $C(O)$.

It follows that:

$$D(O) = 1 - R(O) \tag{12}$$

As much as \mathbf{R} is diverging from being an identity matrix, i.e., $r_{ii}=1$ and $r_{ij}=0$, $D(O)$ is decreasing, being 0 when all the classes in (O) are equal, i.e., $r_{ii} = 1$ and $r_{ij}=1$, since in this case $R(O)=\mathbf{P}'\mathbf{R}\mathbf{P} = 1$.

The measure of diversity of Rao (1982) is a fuzzy diversity measure because it incorporates the degrees of belonging of each of the S classes to the other classes. This degree of belonging in Rao's formula is given by a measure of dissimilarity that is complement to 1 of a measure of similarity. The idea of considering a similarity matrix as a fuzzy matrix is a consequence of considering the similarities as degrees of belonging to sets (cf. Feoli and Orlóci 2011 and references therein). $\mathbf{\Delta}$ and \mathbf{R} can be calculated with several similarity-dissimilarity functions whose values can be easily transformed in such a way to range between 0 and 1 (cf. Orlóci 1978, Podani 2000).

Actually the original function of Rao (1982) measures only the weighted average dissimilarity between the S classes of $C(O)$ since it assumes the complete homogeneity within each of the S classes. However the diagonal of $\mathbf{\Delta}$ may have values ranging between 0 and 1, i.e the dissimilarity within the classes, in this case the $D(O)$ will be the overall average dissimilarity of (O) .

Formula 9) can be considered as a fuzzification of formula 6), however formula 4) also can be easily fuzzified as shown by Ricotta and Szeidl (2006) by introducing the dissimilarity matrix $\mathbf{\Delta}$ in the formula, e.g.:

$$D(O) = -\sum_i p_i \ln(1 - \sum_{j \neq i} \delta_{ij} p_j) \tag{13}$$

That in case of \mathbf{R} instead of $\mathbf{\Delta}$ may be written as:

$$D(O) = -\sum_i p_i \ln(\sum_{j \neq i} r_{ij} p_j)$$

Both the formula 9) and 13) can be easily generalized in the parametric forms of formulas 1) and 2) as shown by Ricotta and Szeidl (2006) and consequently in the parametric form of formula 3) as shown by Leinster and Cobbold (2012) with the formula:

$$D(O) = \left(\sum p_i (\mathbf{R}p)_i^{q-1} \right)^{\frac{1}{1-q}} \quad 14)$$

where the parameter q , called sensitivity parameter, is analogous to the parameter a in formula 3).

The use of functions 1), 2), 3), 13) and 14) allows to find three consistent measures of diversity that can be called respectively the measure of richness S (the number of classes S or $\ln S$ or $S-1$), the Shannon formula H (namely the entropy of order 1, or $\exp H$) and the concentration or dominance C (the formula of Gini-Simpson or some of its different expression). The S, H, C may be used to construct the diversity profiles that may be useful to compare the diversity of environmental sampling units in a consistent way and taking into account three aspects of diversity measurement.

4.3 Measures of evenness

Evenness is a measure of how much the given quantity Q used for weighing the S classes is uniformly distributed among the classes. In terms of similarity it is a measure of relative heterogeneity of $C(O)$. Evenness depends on the formula used for calculating diversity (Ricotta 2004, Jost 2010), in any case it tends to be 0, when the quantity is concentrated in only one class (no heterogeneity), while it is 1 when the classes are equal in terms of w_i , i.e., when all the p_i are equal (maximal heterogeneity). Usually for a $C(O)$ evenness is given by the ratio between the calculated and the maximal theoretical diversity (Pielou 1975, Ricotta 2004, Jost 2010):

$$E = \frac{D(O)}{D(O)_{\max}} \quad 15)$$

This formula implies that we assume that the minimum diversity is always corresponding to 0, i.e., the null hypothesis is considering that, before the diversity analysis of (O) , there is only one class of events for which it would be unjustifiable to consider a partition $C(O)$ of S classes. In case we assume that there is already a partition and that the minimal diversity is occurring when the vector \mathbf{P} of p_i s components has an element equal to $(T-n+1)/T$ (corresponding to the most abundant class) and $(n-1)$ elements equal to $(1/T)$, i.e., a quantity that allows the classes just to be recorded, the diversity would be never 0 and the evenness should be calculated according to Hill (1973) with:

$$E = \frac{D(O) - D(O)_{\min}}{D(O)_{\max} - D(O)_{\min}} \quad 16)$$

However the assumption of minimum diversity equal to 0 should be more correct since, as already mentioned, at the beginning of a study we should assume that all the events of (O) would be equivalent, i.e., would belong to the same set. In other words at the beginning of a study we do not know how the quantity Q could be partitioned in S classes because S is unknown and it is unknown also the similarity-dissimilarity between the S possible classes.

A graphical way to express the evenness for crispy diversity is the well known curve of Lorenz (Lorenz 1905, Gastwirth 1971), currently used in economy to show the pattern of equality within a population in terms of the income of different groups. The curve is constructed by putting in abscissa the cumulative percentage of income starting from the poorest group (i.e., the cumulative relative abundance of the classes) and in the ordinate the cumulative percentage of the groups (i.e., the cumulative percentage of the classes of the objects). For the two examples A and B, given in paragraph 4.4, the curves are presented in Fig. 2.1 with those corresponding to the situation of the maximal possible dominance for A and for B (max Dom).

According the idea that the minimum diversity should be zero, for formula 4) the evenness is:

$$E = \frac{-\sum_i p_i \ln p_i}{\ln S} \tag{17}$$

for formula 6) the evenness is:

$$E = \frac{1 - \sum_i p_i^2}{1 - \frac{1}{S}} \tag{18}$$

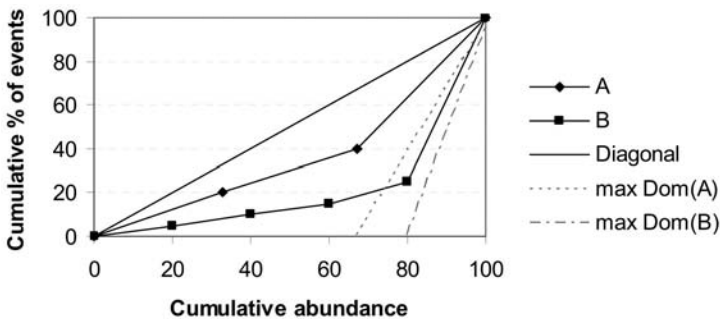


Figure 2.1 The curve of Lorenz for two hypothetical maps A and B.

for formula 9) the evenness is:

$$E = \frac{\sum_i \sum_j \delta_{ij} p_i p_j}{1 - \frac{1}{S}} \quad (19)$$

and for formula 13) the evenness is:

$$E = \frac{-\sum_i p_i \ln (1 - \sum_{j \neq i} \delta_{ij} p_j)}{\ln S} \quad (20)$$

The evenness corresponding to $D(O)$ calculated with formulas 3) and 14) are always obtained by dividing the values of $D(O)$ by S since the maximal diversity for these formulas is always S . In terms of similarity-dissimilarity, S is a measure of total crisp dissimilarity, E is a measure of relative average dissimilarity of the classes of $C(O)$ with respect the maximal crisp dissimilarity.

The evenness is a measure that ranges between 0 and 1, and if it is interpreted as relative average dissimilarity (heterogeneity) its complement may be called relative average similarity (homogeneity) that is a measure of dominance or concentration that in information theory is also known as redundancy:

$$R = 1 - E = \frac{D(O)_{\max} - D(O)}{D(O)_{\max}} \quad (21)$$

4.4 Diversity profiles

With formulas 1), 2), 3), 14) we can draw the diversity profiles of (O) for a given partition $C(O)$. These profiles may be used to compare sampling units in terms of diversity. An example will clarify the idea of diversity profiles. Let us consider two hypothetical maps A and B, corresponding to two environmental sampling units that we want to compare considering the crispy and fuzzy diversity measure. Map A has three classes of events (A,B,F) while map B has 5 classes (A,B,C,D,E). These events may be pedotypes, land cover types, vegetation types etc. that we can call land types including polygons defined at the same hierarchical level (i.e., $S(A)=3$ and $S(B)=5$). The vector $\mathbf{P}(A)$ is composed by $p(A)=0.60$, $p(B)=0.30$, and $p(F)=0.10$ and the vector $\mathbf{P}(B)$ is composed by $p(A)=0.75$, $p(B)=0.10$, $p(C)=0.05$, $p(D)=0.05$, $p(E)=0.05$). The dissimilarities between the classes computed on the basis of a given set of characters (K) may be expressed by the two symmetric matrices of Table 2.1.

Table 2.1 Upper triangular part of the dissimilarity matrices between the classes in A and in B of two hypothetical maps A and B (see the text). The average dissimilarity of A is 0.37, the average dissimilarity of B is 0.495. The evenness of the eigenvalues of the complement of ΔA (Feoli et al. 2009) is 0.65 the evenness of the eigenvalues of the complement of matrix ΔB is 0.69.

ΔA	A	B	F	ΔB	A	B	C	D	E
A	0.00	0.45	0.30	A	0.00	0.45	0.35	0.55	0.55
B		0.00	0.35	B		0.00	0.60	0.35	0.75
F			0.00	C			0.00	0.10	0.65
				D				0.00	0.60
				E					0.00

An example of diversity profiles for A and for B calculated by considering $\alpha \rightarrow 0$, $\alpha \rightarrow 1$ and $\alpha \rightarrow 2$ for formula 1) and $a \rightarrow 0$, $a \rightarrow 1$ and $a \rightarrow 2$ for formula 3) and diversity profiles for formula 13) (Ricotta and Szeidl 2006) and 14) (Leinster and Cobbold 2012) is given in Table 2.2. We do not consider formula 2) since the formulas 1), 3), 13) and 14) also include the formulas of Gini-Simpson.

From Table 2.2 we can draw the graphs of the diversity profiles for map A and for map B, that can be used to have a visual comparison of the two environmental sampling units. One example of such graphs is given in Fig. 2.2.

We do not show profiles considering the parameters α and a bigger than 2 because after such values the curves of different sets (O) are slowly going down almost parallel towards the minimum values of the functions when α and a are tending to infinite. Notwithstanding maps A and B can be very similar in terms of some diversity measures, e.g., Shannon’s formula (H), they are clearly different considering the richness S and the formulas of Gini-Simpson (C) and especially considering the profiles of evenness (E).

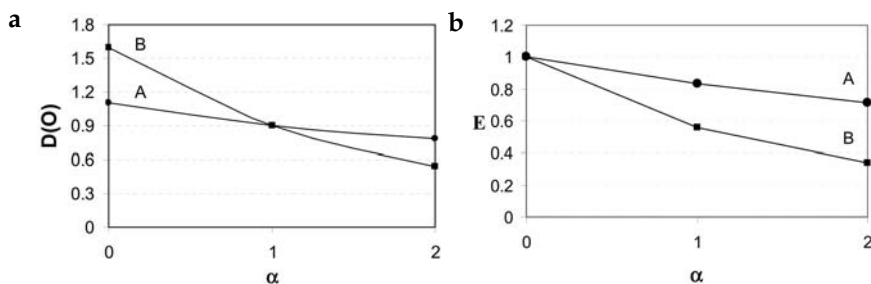


Figure 2.2 Example of diversity profile S,H,C for formula 1 a) and corresponding evenness b) (see Table 2.2).

Table 2.2 Results of the application of formula 1), 3), 13) and 14) with different α and a values to the data of the sampling units A and B. The values for A and B correspond to three points of the diversity profiles (S,H,C) that can be drawn on the basis of crispy diversity, fuzzy diversity and evenness (E).

Crispy-Formula 1)						
α	D(O)		D(O) _{max}		E	
	A	B	A	B	A	B
0	1.10	1.60	1.10	1.61	1.0	1.0
1	0.90	0.90	1.10	1.61	0.83	0.56
2	0.78	0.54	1.10	1.61	0.71	0.34
Crispy-Formula 3)						
a	D(O)		D(O) _{max}		E	
	A	B	A	B	A	B
0	3	5	3	5	1	1
1	2.46	2.46	3	5	0.82	0.49
2	2.17	1.72	3	5	0.72	0.34
Fuzzy-Formula 13)						
α	D(O)		D(O) _{max}		E	
	A	B	A	B	A	B
0	1.10	1.61	1.10	1.61	1	1
1	0.25	0.24	1.10	1.61	0.23	0.15
2	0.25	0.22	1.10	1.61	0.23	0.14
Fuzzy-Formula 14)						
a	D(O)		D(O) _{max}		E	
	A	B	A	B	A	B
0	3	5	3	5	1	1
1	1.28	1.35	3	5	0.43	0.27
2	1.28	1.25	3	5	0.43	0.25

Both the differences between the crispy measures of Shannon and Gini-Simpson can be statistically tested respectively with the formulas suggested by Hutcheson (1970) and Lyons and Hutchenson (1978). The tests for differences between the fuzzy measures have not been explored as yet, however the use of permutation techniques (see Manly 1997) may solve the problem.

4.5 Other ways to express the parametric measures of fuzzy diversity

The most used formulas for crispy diversity obtained from the generalized functions 1), 2) and 3) are respectively 4) and 6) and 7) and 8) while for fuzzy diversity it is used mainly formula 9). Notwithstanding 13) and

14) are promising they may have problems with the similarity functions to incorporate in such formulas (Pavoine et al. 2005, Ricotta and Szeidl 2006, Rao 2010), furthermore when we want to obtain a single number for measuring diversity and not a graph, the formulas known as Gini-Simpson diversity are low sensitive to the number of classes and highly sensitive to dominance, i.e., sets (O) with very different number of classes may have very similar values when only one or few classes are dominant, while the formula known as Shannon index is more sensitive to the number of classes (S) irrespective of their weight. The formula 7) suggested by Hill (1973) has the advantage of being equal to S when there is equal weight among the classes and being less than S as far as the evenness decreases.

Feoli et al. (1992), Feoli and Zuccarello (1996) and Feoli (2010) propose different ways to consider similarity in the diversity measures (fuzzification of diversity) in order to give more freedom to the choice of similarity functions and without incorporating them directly into the formulas as in the case of 9), 13) and 14).

The proposal of Feoli et al. (1992) was to calculate the vegetation diversity of a landscape cover in terms of vegetation types by the following formula:

$$D(O) = H(1 - R) \tag{22}$$

where H is the entropy (formula 4 or formula 6) of the S vegetation types based on their proportional land cover extent and R is the average similarity between them in terms of species composition. In the application of Feoli et al. (1992) it was used the probabilistic similarity index of Goodall (Goodall 1966, Goodall and Feoli 1988, Goodall et al. 1987).

The evenness of 22) is:

$$E = \frac{H(1 - R)}{\ln S} \tag{23}$$

in case H is obtained with formula 4), while it is:

$$E = \frac{H(1 - R)}{1 - \frac{1}{S}} \tag{24}$$

in case H is obtained with formula 6).

The proposal of Feoli (2010) was addressed to measure the niche diversity of a species by the formula

$$D(O) = S^{1-R} \tag{25}$$

Where S is the number of different habitats of the species and R is the average similarity between the habitats. In this formula the p_i , namely the “probability” to find the species in the i th habitat is not considered, however the formula may be changed considering the following:

$$D(O) = S \frac{H(1-R)}{H_{\max}} \quad (26)$$

Where H is obtained by formula 4) or 6) and H_{\max} is $\ln S$ or respectively $1-1/S$. The idea of using a power function of the type $y=x^\phi$, where y is the diversity, x is the number of classes S and ϕ is a number ranging between 0 and 1, is appealing since in case of $\phi=1$, the diversity is equal to S , namely the N_0 of Hill (1973) and in case of $\phi=0$ it is equal to 1. This formula assumes that diversity is firstly related to S and then to some other parameter that accounts for the relationships between the S classes. In case of crispy diversity formula 26) is equal to:

$$D(O) = S \frac{H}{H_{\max}} \quad (27)$$

It is easy to show that formula 27) is another way to write the $N1$ of Hill (1973) if H is calculated with formula 4) since $S^{H/H_{\max}} = (S^{1/\ln S})^H = e^H$.

It follows that formula 26) for H calculated with formula 4) is a fuzzification of formula 13) of Ricotta and Szeidl (2006) and therefore just another way to write it.

The application of formulas 22) and 26) to the maps A and B gives the following results:

22) based on Shannon is 0.33 for A and 0.445 for B, 22) based on Gini is 0.20 for A and 0.21 for B; 26) based on Shannon is 1.39 for A and 1.56 for B while 26) based on Gini is 1.39 for A and 1.52 for B. If we consider the evenness of the eigenvalues of the similarity matrices of A and B as suggested by Feoli et al. (2009), the diversity of A is 2.04 and the diversity of B is 3.03. Based on these formulas the fuzzy diversity of B is always higher than the fuzzy diversity of A. Considering the crispy diversity calculated with formula 27) applied with H of formula 4) diversity is 2.4 for both, while applied with H of formula 6) is 2.46 for A and 2.34 for B.

5. Non Parametric Measures of Diversity

There are several other formulas proposed for measuring diversity that are not related to formulas 1), 2) and 3) that could be used for comparing diversities of environmental sampling units (Magurran 1988, 2004). One

is the formula of rarefaction proposed by Hurlbert (1971) and applied frequently in landscape ecology (Chiarucci et al. 2009), that would give for a given map with S land types and T pixels (quantity Q , that for non-raster maps is the total area T expressed in some standard units, e.g., square meters, hectares, square km etc.), a curve of the richness of land types in function of increasing number of pixels n from 0 to T :

$$D(O) = \sum_i \left(1 - \frac{\binom{T - n_i}{n}}{\binom{T}{n}} \right) \quad i=1, \dots, S \quad (28)$$

the resulting curves are called rarefaction curves, they can be used to compare maps with different numbers of pixels of the same size (specimens) at a fixed number n of them. The map with higher diversity is the one that at parity of pixels would have more land types.

Figure 2.3 presents the rarefaction curves for the two hypothetical maps A and B.

Other measures $D(O)$ used in ecology but that can be used also in other disciplines can be obtained by the formulas of Margalef (1957) and the McIntosh (1967) that are written respectively as:

$$D(O) = \frac{S - 1}{\ln T} \quad (29)$$

And:

$$D(O) = \frac{T - \sqrt{\sum_i n_i^2}}{T - \sqrt{T}} \quad (30)$$

with T equal to the total of quantity Q and $\sum_i n_i = T$.

If we consider the T in maps A and B equal to 100, the diversity of map A and B according to formula 28) will be respectively 0.435 and 0.86, while according to formula 29) will be respectively 0.357 and 0.26. These are crispy measures that may be fuzzified by multiplying them by the average dissimilarity values between the land types of the two maps as it is done for entropy in formula 22). In this case the values will be for formula 28) respectively 0.16 and 0.43 while for formula 29) respectively 0.13 and 0.13. From these scores we can see again that a single number expressing diversity may give contradictory results.

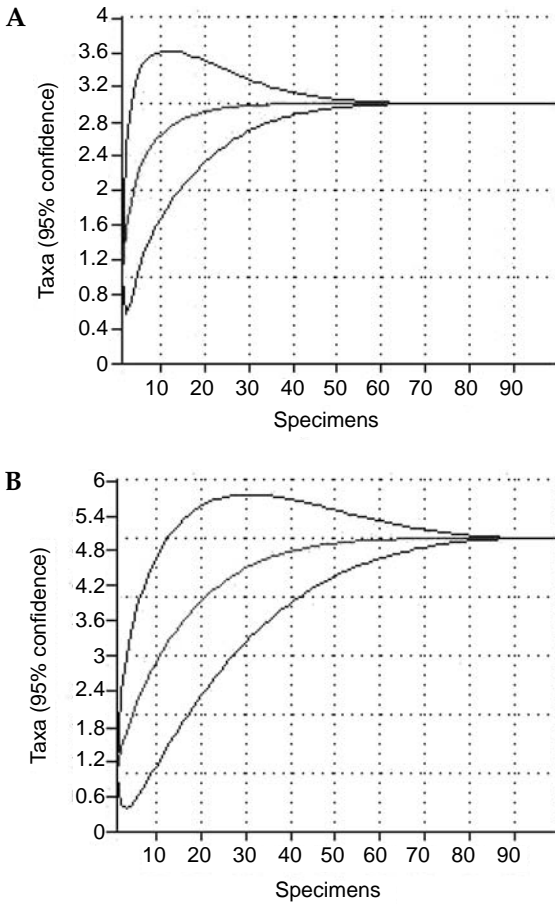


Figure 2.3 Rarefaction curves obtained with the PAST program (Hammer et al. 2001). The specimens indicate the n standardized sample sizes fixed by the user. A = the curve for the map A, B = the curve for the map B.

6. Hierarchical Diversity and Hierarchical Diversity Profiles

The diversity profiles obtained by the formulas 1), 2), 3), 13) and 14) are considering only one partition $C(O)$ at a time, however they can be calculated for comparing the diversity of several partitions at the same hierarchical level, or partitions defined at different hierarchical levels. This means that each class of the S classes of $C(O)$ can be included in a class with other classes of the S classes in a hierarchical way, i.e., by respecting the

principle of inclusion of set theory. In biology the species may be included in genera, the genera in families, the families in orders etc. (e.g., Feoli and Scimone 1984) or the species may be included in functional type of the first hierarchical level that are grouped in functional types of the second hierarchical level and so forth (e.g., Feoli 1984). The idea that the diversity should be analyzed in a hierarchical way was first suggested by Pielou (1975). She justifies her idea by considering that given two sampling units with the same number of classes S , the diversity should be higher for the unit for which the number of classes in which we can group the S classes is higher. In case of two sampling units with different S , as A and B, the diversity could be higher for B than for A only if the classification of the 5 classes would give a total number of classes at different hierarchical levels higher than that of A. Table 2.3 may explain the point. In this table the S classes are assigned to classes of higher hierarchical level. At maximum the number of classes of the hierarchical level $l+1$ can be equal to the number of classes at level l . If for all levels the number of the classes is equal to the number of classes at level 1, then the total number of classes will be $v * S$, where v is the number of hierarchical levels. The matrix of Table 2.3a

Table 2.3 a) Distribution of the 5 classes A, B, C, D, E of map B into classes at three higher hierarchical levels. In total the taxonomic levels starting from the bottom are 4. The classes at the second level are 4, the classes at the third level are 3 the classes at the fourth level are 2. The last row gives the totals of abundance values for the classes at the 3 hierarchical level above the bottom level (A=75, B=10, C=5, D=5, E=5). b) matrix of taxonomic similarity between the five objects.

a)

	Level 2				Level 3			Level 4	
	1	2	3	4	1	2	3	1	2
A	1	0	0	0	1	0	0	1	0
B	1	0	0	0	1	0	0	1	0
C	0	1	0	0	1	0	0	1	0
D	0	0	1	0	0	1	0	0	1
E	0	0	0	1	0	0	1	0	1
n_i	85	5	5	5	90	5	5	90	10

b)

	A	B	C	D	E
A	1	1	0,5	0	0
B	1	1	0,5	0	0
C	0,5	0,5	1	0	0
D	0	0	0	1	0,2
E	0	0	0	0,2	1

can be used to calculate the taxonomic similarity-dissimilarity matrix (\mathbf{R} or Δ) of the S classes, that can be used in formulas 9), 13) and 14) to calculate the hierarchical diversity. We thus define the hierarchical diversity of a $C(O)$ as a fuzzy diversity that takes into consideration the similarity of the classes according to their hierarchical arrangement. The crispy diversity of a $C(O)$ can be decomposed into specific components for each hierarchical level in which the classes are grouped (Pielou 1975, Feoli and Scimone 1984, Legendre and Legendre 1998), but this would not correspond to the concept of hierarchical diversity, this is just the process of decomposing the diversity of $C(O)$ in its hierarchical components.

The evenness of the eigenvalues of the taxonomic similarity in Table 2.3a, according to Feoli et al. (2009) is 0.78 therefore the hierarchical diversity calculated with the general formula $y=x^e$ is $D(B)=3.5$.

An example of constructing hierarchical profiles, in a situation where there is not a given taxonomy on which to base the aggregations of the events (O) at specific hierarchical levels can clarify what the hierarchical diversity profiles are in a more direct way than that shown by Feoli (2012). Let us consider an area for which a sample of 16 points (or subareas of the same size) is sampled for calculating the diversity of the area with respect to some environmental system that should be mapped (vegetation, soil, or land cover, etc.) (Fig. 2.4a). Let us consider the dendrogram of Fig. 2.4b obtained from the similarity matrix of Table 2.5a that has been obtained from a matrix describing the 16 points on the basis of some characters of the system (e.g., soil variables if the system is the soil).

The dendrogram can be used to define hierarchical levels at which to compute the diversity. The simple visual inspection of the dendrogram structure suggests three main levels. At the first level we have five clusters, at the second level four and at the third level three. The frequency distributions of the points in the clusters can be used to calculate the crispy diversity of the map, while with the resemblance matrices within and between the clusters at the three levels we can calculate the fuzzy diversity and the evenness on the basis of Shannon or Gini-Simpson index (Ricotta and Szeidl 2006) and Rao (1982). The results are given in Table 2.4. The evenness of the eigenvalues $E(\lambda)$ (Feoli et al 2009) for the matrices of similarity between the clusters is also reported. The value of $E(\lambda)$ reaches the highest value at three clusters. If we consider two clusters the value goes down again to a value around 0.72 owing to a significant increment of the heterogeneity within

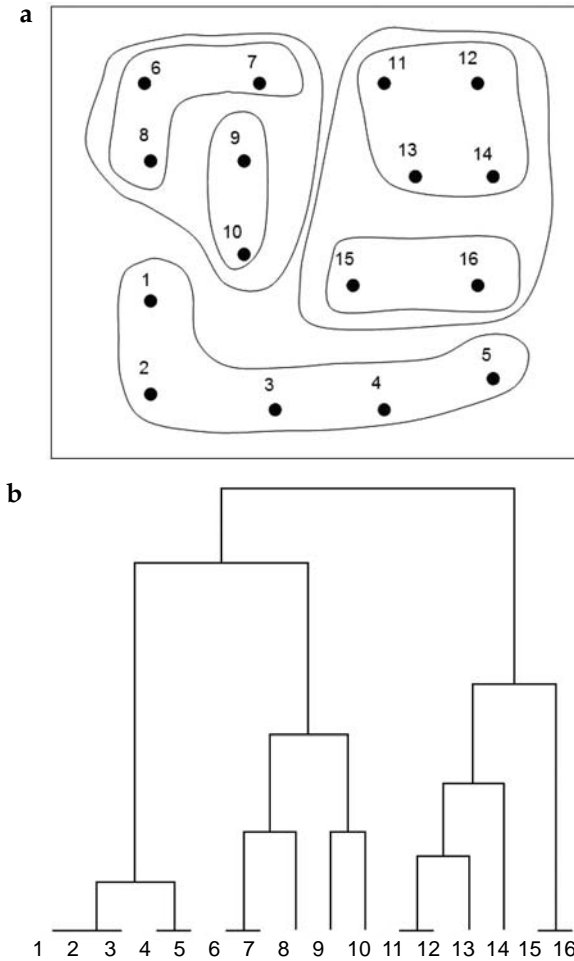


Figure 2.4 a) An hypothetical map representing an environmental sampling unit with 16 objects corresponding to a sample done for describing the unit, and b) the dendrogram showing the similarity between the objects describing the sampling unit according to a matrix of k characters.

the clusters (Table 2.5b). Figure 2.5 shows an example of the hierarchical crispy diversity profiles in function of the parameter α and a .

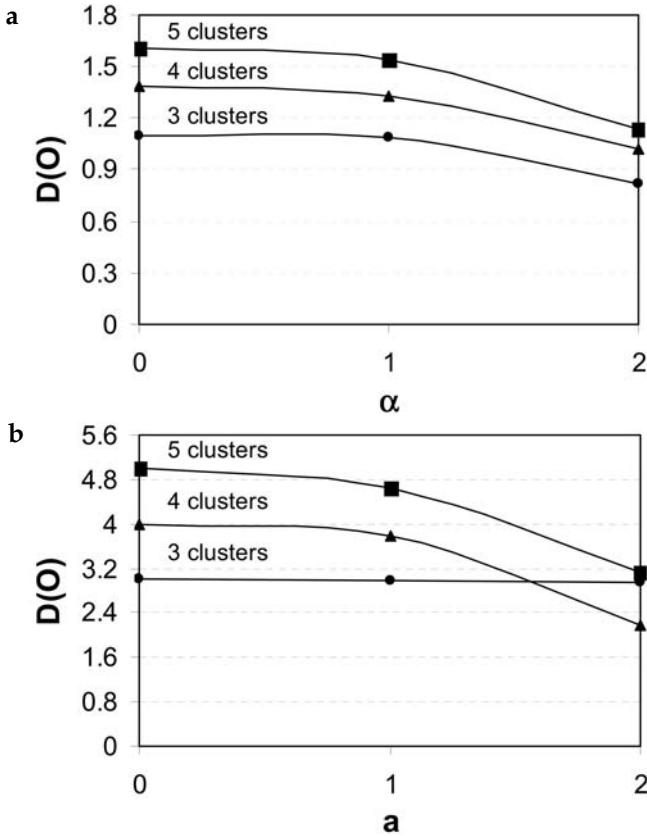


Figure 2.5. Examples of hierarchical diversity profiles for data in Table 2.4 for parameter α e parameter a .

7. Discussion and Conclusions

We have presented a short review of the formulas for measuring diversity of a sampling unit extracted from the environmental system with examples related to maps of land types. These kind of examples have been chosen because pedodiversity would be mainly calculated on the basis of such maps. We have defined diversity under the perspective of similarity-dissimilarity between the objects describing the sampling units. It has been shown that under this perspective the diversity measured by the parametric functions of Renyi (1961) and Patil and Taillie (1976) are just measures of weighted similarity-dissimilarity between the objects. We have suggested that the similarity matrix $\mathbf{R}(T \times T)$ (with T the total Q

Table 2.4 a) Measures of diversity (Shannon and Gini) and different measures based on evenness (E) for the clusters at three hierarchical levels.

Clusters	E(λ)	Frequencies in the clusters	Shannon		Gini		E _{Shannon}		E _{Gini}	
			Crispy	Fuzzy	Crispy	Fuzzy	Crispy	Fuzzy	Crispy	Fuzzy
3	0.81	5 5 6	1.10	0.58	0.66	0.45	0.99	0.53	0.99	0.67
4	0.79	5 5 4 2	1.33	0.64	0.73	0.47	0.96	0.46	0.97	0.63
5	0.72	5 3 2 4 2	1.54	0.68	0.77	0.49	0.91	0.42	0.96	0.61

Table 2.4 b) Diversity measures according to the formula $y=x^\phi$ where ϕ is the evenness E, as in formulas 26) and 27), given in Table 2.4a for fuzzy diversity.

Clusters	E(λ)	E _{Shannon}	E _{Gini}
3	2.44	1.79	2.08
4	2.99	1.89	2.39
5	3.18	1.96	2.67

Table 2.5a Matrix of similarity between the 16 objects of Fig. 2.4a. The evenness of the eigenvalue of this matrix (Feoli et al. 2009) is 0.58.

Matrix of similarity	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
1	1	0.95	0.95	0.84	0.84	0.32	0.32	0.42	0.53	0.63	0.32	0.32	0.42	0.53	0.53	0.53
2	0.95	1	0.95	0.89	0.89	0.32	0.32	0.42	0.53	0.63	0.32	0.32	0.42	0.53	0.53	0.53
3	0.95	0.95	1	0.89	0.89	0.32	0.32	0.42	0.53	0.63	0.32	0.32	0.42	0.53	0.53	0.53
4	0.84	0.89	0.89	1	0.95	0.16	0.16	0.26	0.37	0.47	0.16	0.16	0.32	0.37	0.37	0.37
5	0.84	0.89	0.89	0.95	1	0.16	0.16	0.26	0.37	0.47	0.16	0.16	0.32	0.37	0.37	0.37
6	0.32	0.32	0.32	0.16	0.16	1	0.95	0.74	0.63	0.53	0	0	0.11	0.26	0.21	0.21
7	0.32	0.32	0.32	0.16	0.16	0.95	1	0.74	0.63	0.53	0	0	0.11	0.26	0.21	0.21
8	0.42	0.42	0.42	0.26	0.26	0.74	0.74	1	0.74	0.63	0.11	0.11	0.21	0.32	0.32	0.32
9	0.53	0.53	0.53	0.37	0.37	0.63	0.63	0.74	1	0.74	0.21	0.21	0.37	0.42	0.42	0.42
10	0.63	0.63	0.63	0.47	0.47	0.53	0.53	0.63	0.74	1	0.32	0.32	0.42	0.53	0.53	0.53
11	0.32	0.32	0.32	0.16	0.16	0	0	0.11	0.21	0.32	1	0.95	0.79	0.63	0.42	0.42
12	0.32	0.32	0.32	0.16	0.16	0	0	0.11	0.21	0.32	0.95	1	0.79	0.63	0.42	0.42
13	0.42	0.42	0.42	0.32	0.32	0.11	0.11	0.21	0.37	0.42	0.79	0.79	1	0.74	0.53	0.53
14	0.53	0.53	0.53	0.37	0.37	0.26	0.26	0.32	0.42	0.53	0.63	0.63	0.74	1	0.63	0.63
15	0.53	0.53	0.53	0.37	0.37	0.26	0.26	0.32	0.42	0.53	0.42	0.42	0.53	0.63	1	0.95
16	0.53	0.53	0.53	0.37	0.37	0.26	0.26	0.32	0.42	0.53	0.42	0.42	0.53	0.63	0.95	1

Table 2.5b Matrices of similarity between the clusters corresponding to four hierarchical levels of the dendrogram of Fig. 2.4b. The evenness of the eigenvalues of these matrices (Feoli et al. 2009) are respectively 0.72, 0.79, 0.81, 0.72.

1st level	1	2	3	4	5	2nd level	1	2	3	4	3rd level	1	2	3	4th level	1	2
1	0.91	0.29	0.52	0.34	0.46	1	0.91	0.38	0.34	0.46	1	0.91	0.38	0.38	1	0.56	0.32
2	0.29	0.81	0.61	0.12	0.25	2	0.38	0.68	0.21	0.34	2	0.38	0.68	0.25	2	0.32	0.63
3	0.52	0.61	0.74	0.35	0.47	3	0.34	0.21	0.75	0.5	3	0.38	0.26	0.63			
4	0.34	0.12	0.35	0.75	0.5	4	0.46	0.36	0.5	0.95							
5	0.46	0.28	0.47	0.5	0.95												

expressed by a natural number) in which the n_i objects of one class define a submatrix $\mathbf{R}_i(n_i \times n_i)$ is the basis to calculate the diversity of a partition $C(O)$. According to this view, when $\mathbf{R}(T \times T)$ is a crispy matrix, i.e., the submatrices are completely disjointed (complete similarity within the S classes of the objects and zero similarity between the classes) the n_i are the eigenvalues of $\mathbf{R}(T \times T)$. In this situation the evenness of these eigenvalues is equal to the formulas 17) and 18) that are expressing the relative measure of heterogeneity of $\mathbf{R}(T \times T)$. It is easy to prove that when $\mathbf{R}(T \times T)$ is not a crispy matrix the evenness of the eigenvalues of $\mathbf{R}(T \times T)$ is lower than that of the corresponding crispy matrix (Wilkinson 1965, Noy-Meir 1973, Feoli 1977). We have also shown that the use of only one number to measure the diversity of the $C(O)$ s that have to be compared may be problematic because different situations in richness and heterogeneity (or dominance-concentration), may show the same value of diversity. This may be acceptable if we are not interested in knowing by what parameter of the diversity components (richness or heterogeneity) the $C(O)$ s are different. In any case it is our opinion that richness is always the component that would be considered more important in discriminating the $C(O)$ s in terms of diversity. We think that S should be the basic number of reference for diversity comparisons according to Hill (1973). We therefore suggest as a reasonable index of diversity the linear $y = \phi x$ or power $y = x^\phi$ combination of richness (x) and ϕ (relative average heterogeneity ranging between 0 and 1) depending on specific consideration of the researcher. However we suggest representing the diversity of a partition $C(O)$ always by the three numbers corresponding to S , H and C (a diversity profile) that could be calculated by the formula the researcher prefers in definite circumstances and by evenness profiles.

At the beginning of this chapter it was mentioned that diversity is a controversial concept that needs a clear definition before starting to measure it. We conclude that measuring diversity can be controversial too, as the utility of the measures depends on the framework in which we want to do them and because several measures of diversity can be easily invented. First of all we have to decide if we want to measure diversity in a crispy or in a fuzzy context, then we have to decide how important are the number of classes with respect to the average heterogeneity of the classes. From the pragmatic point of view, what is promising and encouraging is the fact that using measures of diversity general trends have been discovered that are common for different disciplines. For example in the study of biodiversity and pedodiversity it was found the same type of richness-area curves (Ibanez et al. 1990) and also that extreme environmental situations show lower diversity both for vegetation types (e.g., Feoli and Bressan 1972, Feoli and Lagonegro 1982) and for pedotaxa (Minasny et al. 2010). Three different

measures of diversity should be necessary for meaningful comparisons in order to leave out ambiguity. What is mathematically interesting is the fact that these measures may belong to the same generalized formulas. By considering the limits of the parameter of the formulas tending to zero, one and two, the resulting diversity measure is corresponding respectively to a measure of richness, to a measure of entropy and to a measure of average heterogeneity (complement of dominance-concentration or average similarity). For example if we are interested in comparing map A with map B in terms of diversity we can use the numbers expressing the diversity (i.e., richness, entropy and heterogeneity) as those obtained in Table 2.2. On the basis of Fig. 2.2 we can see that map A and B are different for richness (number of different classes) and for Gini-Simpson index (that is sensitive to dominance), while they are equal for entropy (Shannon index), this means that the evenness in A is compensating its lower richness. The comparison of few sampling units and the values of a unit described at different hierarchical levels, are feasible with diversity profiles when the partitions $C(O)$ s to be compared are few, however when we want to compare several sampling units then we have to use the similarity between the diversity profiles and cluster analysis and ordination methods to detect the mutual position of the sampling units in the dimensional space defined by the three measures of diversity.

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CHAPTER 3

Nonlinear Dynamics, Divergent Evolution, and Pedodiversity

Jonathan D. Phillips

1. Introduction

Diversity and variability of soils may arise from extrinsic or intrinsic factors. Extrinsic factors are external to the soil itself, and represent environmental controls such as the soil-forming factors of climate, biota, topography, parent material and external disturbances, such as human agency, hydrometeorological events, volcanic eruptions, etc. Intrinsic factors are internal to the soil, and include pedogenetic processes and interactions among soil components. The interaction of intrinsic and extrinsic factors may also influence pedodiversity. Pedogenetic processes may act to reinforce and exaggerate soil variations associated with environmental controls, or to smooth and reduce such variations. The amplification (or filtering) of initial variations and disturbances can result in the divergent (or convergent) evolution of soils, with resulting increases (or decreases) in pedodiversity. This chapter discusses the role of nonlinear dynamics in pedogenesis with respect to these phenomena.

Soil differences directly related and proportional to variations in extrinsic factors are straightforward to understand and explain (though typically not simple or easy, due to difficulties in measuring variability of environmental controls and disturbances). Differences in soils arising

from intrinsic controls, or from amplification of small variations in extrinsic factors, are more problematic, as soil variations may not be linked to any observable variations in environmental controls.

Soil formation and evolution may be characterized by dynamical stability or instability. Stable pedogenesis indicates intrinsic processes whereby minor variations in initial conditions or the effects of local disturbances are not magnified over time, and may well be obscured or diminished. Dynamical instability involves dynamics that result in the persistence and growth of (the effects of) small initial variations and local perturbations. In a nonlinear dynamical system (NDS) such as pedological, geomorphological, and ecological systems, dynamical instability is equivalent to deterministic chaos, often conceptualized as variability arising strictly from the intrinsic dynamics of a system, independently of effects of extrinsic controls. Dynamical (in)stability and chaos in pedogenesis is described in detail, with empirical examples, in some earlier papers (Phillips 1993a, 1993b, 1998, 2000, 2001b, Phillips et al. 1996).

We can therefore outline several scenarios with respect to variation of soils relative to that of environmental controls, as shown in Table 3.1. Note that the relative importance of intrinsic and extrinsic factors, and the stability of pedogenetic processes, may vary with spatial and temporal scale within the same soil landscape. For instance, several studies have found that broad-scale variations in the suite of soils found within a given landscape are well explained by observed variations in environmental controls, with local variations dominated by intrinsic instabilities (Phillips 2001a, Phillips and Marion 2005). Divergent evolution refers to increasing differentiation of soils; convergent to increasing spatial homogenization of soils over time.

A given Earth surface system may be dynamically stable and convergent; or unstable, chaotic, and divergent in various situations and at different temporal or spatial scales (Phillips 1999). This chapter will focus on instability and divergence as a source of pedodiversity, and on pedodiversity at the landscape scale. Earlier publications explicitly linking dynamical instability and chaos to pedodiversity include Phillips (2001a, 2001b), Caniego et al. (2007), Saldana and Ibáñez (2007), and Borujen et al. (2010).

Table 3.1 Soil variation related to that of extrinsic soil-forming factors (SFF).

Soil variation dominated by . . .	Intrinsic processes dominantly . . .	Soil development	Variation relative to SFF
Extrinsic factors	Stable	Convergent	Less than
Extrinsic factors	Unstable	Divergent	Greater than or equal to
Intrinsic factors	Stable	Convergent	Less than or equal to
Intrinsic factors	Unstable	Divergent	Greater than

2. Background

2.1 Pedodiversity and soil variation

As discussed in more detail elsewhere in this volume, pedodiversity consists of at least three aspects: richness (the number of different soils), differentiation (the degree of difference among the soils) and evenness (relative abundance of soil types). Soil variation includes the three aspects of pedodiversity, plus the spatial pattern of soil types.

Like other environmental phenomena, soils may have sharp boundaries and be readily distinguished from one another, or may vary continuously, with only gradations between different soil types. Both phenomena (and of course intermediate cases) are possible in the same landscape with respect to different soil properties or features. As in biological taxonomy, geological mapping, and classification and mapping of landforms and vegetation communities, boundaries (in both the geographical and taxonomic sense) may sometimes be arbitrary. Acknowledging that continuous representations of soils are sometimes more appropriate, this chapter accepts the notion of soils as discrete entities—that is, there exist fundamentally different soil types.

This synthesis is also based on a factorial model of soil formation and soil geography, whereby soils are products of the environment, and soils at a given location are a function of the combined, interacting effects of (at least) geology, climate, topography, biota, and the age of the surface on which the soils are formed. This is, of course, consistent with the familiar “clorpt” model. The environmental controls and influences of soil are referred to here as soil forming factors (SFF).

In general, we expect pedodiversity to be at least roughly proportional to the diversity of the SFF. However, stable, convergent pedogenesis may mask or obscure some variations of the SFF. Further, in some cases a single state factor is dominant—if this is the case, and the dominant factor is less spatially variable than other SFFs, then pedodiversity may be less than overall state factor diversity.

Conversely, divergent, unstable pedogenesis may amplify initial variations in SFF, and superimpose additional variations due to localized disturbances (Fig. 3.1). Assuming that increased variability of soil properties leads at some point to differentiation of soil types, this indicates pedodiversity greater than SFF diversity.

For soils to be more diverse than their parent material, at least one of the following situations must hold:

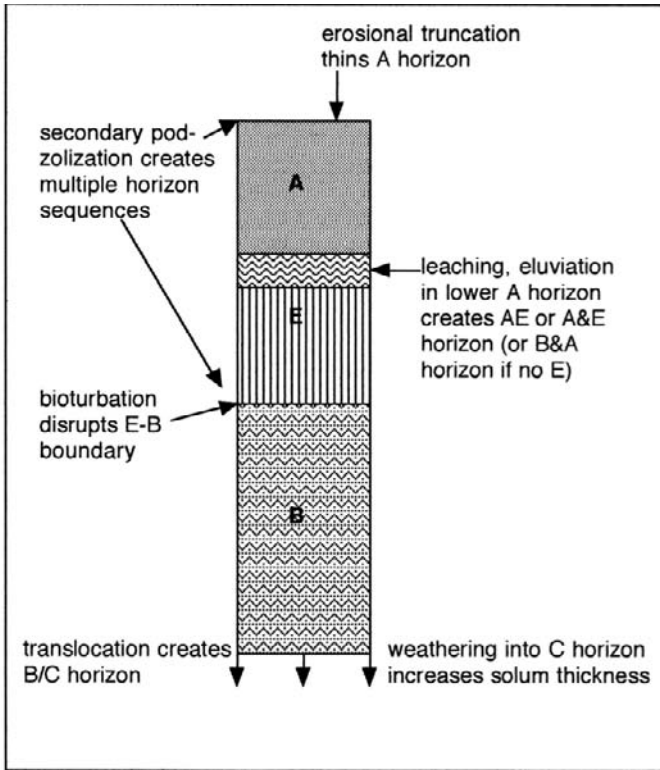


Figure 3.1 Diagram showing pedogenetic processes and their effects on soil morphology. In the case study from which this is taken (Phillips 2000), the net effect is to increase the entropy and morphological diversity of the soil over time.

- One or more other state factors is more diverse than parent material.
- Pedologic effects of local disturbances cause variations within similar parent materials.
- Pedogenesis is dynamically unstable, chaotic, and divergent.

The traditional SFFs are climate, organisms or biota, relief or topography, parent material and time. Climate includes a regional component, superimposed with local variability. The latter is typically associated with topography through factors such as aspect, elevation, and topographically driven or influenced water and air flows (though land surface characteristics associated with vegetation cover, human modifications, and soil or parent material properties themselves may also play a role). Climate, then, is unlikely to be more variable than topography. Any variations in the time factor independent of parent material are likely to be associated with disturbances.

This suggests that where pedodiversity is greater than the diversity of both parent material and topography, the diversity is associated with biotic effects, local disturbances or divergent pedogenesis (or some combination of these).

3. Soils as Nonlinear Dynamical Systems

3.1 Soils are nonlinear systems

In linear systems, outputs or responses are proportional to inputs or stimuli across the entire range of the latter. In nonlinear systems this is not the case. Soil processes are nonlinear because they are characterized by thresholds, and thresholds by definition mean that there is disproportionality between inputs and outputs. A number of authors explicitly discuss pedogenetic thresholds (e.g., Muhs 1984, Chadwick and Chorover 2001, Ewing et al. 2006, Phillips 2006b, Zehe and Sivapalan 2009), but most texts on pedology and soil geomorphology describe numerous threshold phenomena (e.g., Ollier and Pain 1996, Birkeland 1999, Schaetzl and Anderson 2005). Other common sources of nonlinearity in soils are storage effects, saturation and depletion phenomena, positive feedbacks, self-limiting processes, competitive relationships, multiple modes of adjustment and several types of self-organizing patterns (Phillips 2003).

Nonlinear systems are not all, or always, complex, and complicated systems are not all nonlinear. However, some complex phenomena, such as deterministic chaos, may occur in nonlinear systems that are not possible in linear systems. Chaos is defined as sensitivity to initial conditions or to small perturbations, such that the effects of either tend, on average, to grow over some finite time. While chaos is usually considered in the temporal domain, in the presence of anything other than perfect spatial isotropy, temporal chaos must lead to spatial-domain chaos (Phillips 1993c). In a pedological context, chaos implies divergent evolution, and that effects of variations in initial conditions or local disturbances are disproportionately large.

Most work on chaos theory and dynamical instability has been in the context of numerical modeling (for pedological examples, see Phillips 1993c, Minasny and McBratney 1999, 2001, D'Odorico 2000, Furbish and Fagherazzi 2001, Caruso and Rillig 2011). Perhaps for this reason, the heritage of nonlinear dynamical systems analysis in mathematics, physics and systems theory, and the popularity of metaphors such as the butterfly effect, many Earth and environmental scientists have not seen complex nonlinear dynamics as relevant to historical and field-based science. Further, in pedology and related fields, it is often impossible to observe or reliably estimate variations in initial conditions or past disturbances. Nonetheless, chaotic phenomena (though often not recognized as such) such as increasing

irregularity over time, high levels of spatial variability uncorrelated to observable environmental controls, and divergent evolution have long been well known in pedology and geomorphology. Earlier work (Phillips 1999, 2006a) explicitly and formally made those connections, as well as suggesting analytical methods appropriate for field-based sciences.

As an example, consider small local variations (i.e., small joints) in the bedrock forming the parent material for a forest soil. Plant roots enter these joints and enlarge them by focusing water flow and biological activity, facilitating weathering. These become locally deeper pockets of soil, which are preferentially exploited by tree roots, further deepening the soil and increasing the spatial heterogeneity of soil thickness. Thus minor variations in initial conditions grow larger over time, and soil thickness (and likely other properties) becomes increasingly variable—that is, dynamically unstable and chaotic. These phenomena are described and empirical verification presented elsewhere (Phillips and Marion 2004, Phillips 2008). Chaotic divergence cannot continue indefinitely, and in this case would presumably terminate as soil depths approach the rooting depth of trees.

3.2 Instability and chaos

Soils and soil-forming factors, to varying degrees, influence and are influenced by, each other. Pedologic systems are thus represented as consisting of n components x_i ($i = 1, 2, \dots, n$), described by a series of nonlinear differential equations:

$$dx_i/dt = f(dx/dt)$$

where x is the vector of all x_i . The soil system state at time t is:

$$x(t) = C x(0) e^{\lambda t}$$

where $x(0)$ indicates the initial system state, and C a vector constant. The λ are the n Lyapunov exponents of the system, equal to the real parts of the complex eigenvalues of the Jacobian matrix of the equation system. The Lyapunov exponents have the property $\lambda_1 > \lambda_2 > \dots > \lambda_n$. Dynamical instability and chaos is present if any eigenvalue is positive, and is thus indicated by $\lambda_1 > 0$.

The mean difference between randomly selected pairs of locations in a soil landscape at t , in terms of an indicator of soil system state such as thickness, surficial sand content, pH, etc. is

$$\delta(t) = k e^{\lambda t}$$

The constant k normalizes the initial separation. Rewriting as follows shows that the largest Lyapunov exponent (and thus whether the system is chaotic) can be determined by analysis of mean divergence (or convergence) over time:

$$\lambda_1 = \ln \delta(t) - \ln k$$

Kolmogorov (K-) entropy of a NDS is equal to the sum of the positive Lyapunov exponents. Thus finite, positive K-entropy indicates dynamical instability and chaos (K-entropy < 0 in a stable, convergent NDS and is infinite in a random system). K-entropy in turn can be calculated as the change over time in Shannon or information entropy, a common tool in the analysis of pedodiversity and soil spatial patterns (e.g., Culling 1988, Ibáñez et al. 1990, Lark 2001, Caniego et al. 2006, 2007).

The direct links between convergence/divergence or entropy changes and stability of NDS are the basis for many of the practical methods for assessing chaos in Earth surface systems (Phillips 2006a).

3.3 Chaos and instability in soils

There now exists extensive empirical (as well as theoretical and model) evidence of dynamical instability and chaos in the formation and evolution of regoliths and soils. Space does not permit a full review, but Table 3.2 summarizes some selected studies. Chaos and instability is directly related to pedodiversity, as it results in progressive differentiation of the soil cover over time, independent of variation in SFF.

3.4 Pedodiversity and instability in forest soils

An example of processes and controls that exhibit instabilities that contribute to pedodiversity is the effect of individual trees on forest soils in the Ouachita Mountains, Arkansas, USA. The Ouachitas are not high elevation (peaks are < 500 , and often < 400 masl), but do have high relief and steep slopes. Parent material for residual soils is sedimentary rock comprised of interbedded sandstones and shales, with some quartz and other lithologies. The rocks are strongly contorted by tectonic stresses, and local lithological and structural variation is an important source of soil variability and diversity.

After noting various field indications that trees repeatedly occupy the same microsites, Phillips and Marion (2004) developed the self-reinforcing pedologic influences of trees (SRPIT) conceptual model, suggesting that over multiple generations of forest, the soil cover diverges into

Table 3.2 Studies showing empirical evidence of dynamical instability, chaos, or divergent evolution of soils.

Location/environmental setting	Evidence of chaos or divergent evolution	Reference
Coastal dune soils, Queensland, Australia	Increasing variability over time of depth to B-horizons	Thompson 1983, 1992
Fluvially-dissected Mediterranean landscapes, Spain & Portugal	Increasing spatial entropy of soil pattern over time	Ibáñez et al. 1990, Ibáñez 1994
Coastal marsh soils, USA Atlantic and Gulf coasts	Sensitivity to small perturbations; divergent evolution	Orson and Howes 1992, Nyman et al. 1993, Hackney et al. 1996
North Carolina coastal plain uplands, USA	Increasing soil richness in a chronosequence	Phillips 1993a
Sandy lake terraces, Michigan, USA	Increasing soil richness and complexity in a chronosequence	Barrett and Schaetzl 1993
Podzolized forest soils, Canada	Disproportionately large microtopographically induced variations in soil morphology	Price 1994
Semi-arid soils, Botswana	Disproportionately large microtopographically induced variations in soil morphology	Miller et al. 1994
Drylands, New Mexico, USA	Divergent evolution; increasing spatial variability of soil nutrients and carbon over time	Abrahams et al. 1995
Pleistocene dune soils, coastal plain, North Carolina, USA	Spatial variation in soil morphological properties disproportionately large relative to SFF	Phillips et al. 1996
Rio Negro basin, Brazil	Divergent evolution in a single parent material	Dubroeuq and Volkoff 1998
North Carolina coastal plain, USA	Finite positive K-entropy of soil spatial pattern resulting from landscape soil redistribution; high level of intrinsically-controlled soil variability	Phillips et al. 1999, Phillips 2001c
Coastal plain Ultisols, North Carolina, USA	Finite positive K-entropy in soil profile development	Phillips 2000
Strand plain, Michigan, USA	Increasing variability in a chronosequence	Barrett 2001
Upland Ultisols, east Texas, USA	Divergent weathering profile evolution indicated by state probability function	Phillips 2001c

North Carolina coastal plain, USA	Divergent evolution of soil cover, indicated by state probability function	Phillips 2001a
Fluvial terraces, central Spain	Increasing pedodiversity with terrace age	Saldana and Ibáñez 2004
Forest soils, Ouachita Mountains, Arkansas, USA	Divergent evolution; domination by intrinsic controls of soil variability	Phillips and Marion 2005
Zayandeh-rud Valley, Iran	Divergent development & finite positive K-entropy in soil landscapes	Toomanian et al. 2006
Forest soils, Ouachita Mountains, Arkansas, USA	Dynamical instability in soil thickness associated with effects of individual trees	Phillips 2008
Rice paddy Vertisols, Cuba	Chaotic behavior in spatial transects of soil physical properties	Milan et al. 2009
Borujen region, central Iran	Spatial variation in soil morphological properties disproportionately large relative to SFF; statistical signatures of chaos	Borujeni et al. 2010
Forest soils, Czech Republic	Spatial variation in soil morphological properties disproportionately large relative to SFF; domination by intrinsic controls of soil variability	Samonil et al. 2011

patches repeatedly occupied by trees, and relatively tree-poor patches rarely influenced by trees. The tree-rich patches, they argued, would be systematically deeper in terms of total soil depth to bedrock, due to tree root interaction with the underlying rock, facilitating weathering, and excavating material from the soil-rock interface during tree uprooting. Later an additional mechanism was discovered, the infilling of stump holes (Phillips and Marion 2006). The displacement of rock fragments by tree growth and the nutrient enrichment following tree mortality apparently enhance the favorability of the microsites (Phillips and Marion 2004, 2006).

These phenomena represent instability because the effects of a relatively small and short-lived event (growth and death of a tree) result in impacts disproportionately large and long-lived. The unstable nature of the tree (and lithological) effects on pedodiversity was explored more explicitly by Phillips and Marion (2005) using the richness-area techniques described in the next section and by Phillips et al. (2005b), who showed that the highly local soil variation, dominated by intrinsic factors, is unrelated to local microtopographic variation. Other works explored the specific mechanisms involved in local soil deepening by trees, including rock fragment redistribution, exploitation of bedrock joints by tree roots and associated organisms and locally accelerated weathering in such joints (Phillips et al. 2005a, Phillips et al. 2008a, 2008b). Other field evidence was used to test specific hypotheses regarding root-bedrock interaction (Phillips et al. 2008a, 2008b) and systematically deeper soil underneath trees than in immediately adjacent locations (Phillips 2008). These tests all strongly supported the SRPIT framework and the role of instabilities associated with tree effects in increasing pedodiversity.

The results from the Ouachitas are likely to be applicable to other forest soils, at least where depth to bedrock is less than the maximum rooting depth of dominant trees. They also suggest the possibility of similar phenomena in other environments, particularly where ecological engineering has significant effects on soil morphology.

3.5 Richness-area analysis

Richness-area analysis is an approach to assessing dynamical instability and chaos in soils that is directly related to pedodiversity. Ibáñez et al. (1995, 1998) first applied richness vs. area curves originally developed in biogeography to studies of pedodiversity. Phillips (2001b) developed the method described below specifically to examine the relative importance of extrinsic (SFF) vs. intrinsic (local instabilities) factors in pedodiversity.

Defining S as the number of different soil types (soil richness) and A as area, $S = f(A)$. Previous work suggests that a power function is most appropriate;

$$S = c A^b$$

For accumulative sampling curves where the S vs. A relationship is developed by successively sampling larger or additional areas, the coefficient c represents richness in the smallest areal unit, while the exponent b indicates the rate of increase of soil types with area. For this type of curve (as opposed to a relationship developed by sampling discrete units of varying area), $c > 1$, and $b > 0$.

A can be divided into n homogeneous units A_i , $i = 1, 2, \dots, n$, such that $A = \sum A_i$ and $S_i = c_i A_i^{b_i}$. The A_i are called elementary units, and are constant with respect to SFF. Thus soil variability within the elementary areas must, by definition, be due to intrinsic rather than extrinsic factors. With no intrinsic variability there is a one-to-one correspondence and $c_i = 1$; $b_i = 0$. Therefore c_i reflects inherent diversity associated with unit i , and b_i the tendency for larger areas or more samples of unit i to have increasing pedodiversity independent of environmental heterogeneity related to SFF.

$$S = \sum (c_i A_i^{b_i} - k_i) = m \sum (c_i A_i^{b_i})$$

where k_i is the number of soil types in unit i that have already been counted in other units, and m an adjustment factor for taxas counted in multiple elementary units ($m = S/\sum S_i < 1$). Therefore

$$S = \bar{c}_i \bar{A}_i^{\bar{b}_i} m n$$

with the overbars indicating mean values.

The ratio \bar{b}_i/b therefore provides an indication of the relative importance of extrinsic, between-unit sources of variability, and intrinsic, within-unit variability. If the elementary units indeed represent homogeneous SFF, then the ratio indicates the extent to which pedodiversity is associated with dynamical instability and deterministic chaos. In many cases the number of samples N , N_i can be substituted for A , A_i in any of the equations above. Phillips (2001) also shows how the ratio of the mean value of c_i to c indicates the average inherent diversity of the elementary units relative to the area as a whole.

For an agricultural soil landscape on the North Carolina (USA) coastal plain, Phillips (2001b) found a \bar{b}_i/b ratio of 1.145, indicating a greater

importance for chaotic, intrinsic, within-unit variation in determining pedodiversity. Phillips and Marion (2007) applied the method to forest soils in the Ouachita Mountains to compare pedodiversity based on U.S. Soil Taxonomy to an *ad hoc* soil geomorphology classification developed specifically for the study area. For the soil geomorphology classification $b_i/b = 1$, indicating equal importance of intrinsic and extrinsic variations, while for soil taxonomy the ratio was 1.15. Other applications of the method include Svoray and Shoshany (2004), who used richness-area analysis to examine the role of intrinsic vs. extrinsic sources of variability of soil drying rates, and Ibáñez et al. (2009), who employed it as one of several methods to examine spatial distributions of soils in Europe.

4. Limits to Pedodiversity

If landscape-scale pedodiversity is not (necessarily) limited by diversity of SFFs, is there any limit on divergent pedogenesis other than time? Given the pedon concept, at least one upper limit must be the total available area, since discrete soil types cannot be indefinitely small.

Divergent pedogenesis associated with deterministic chaos and dynamical instability also has limits, first of all because these phenomena by definition are finite. In addition, a common feature of soil and other Earth surface systems is conditional stability—that is, whether the system is dynamically stable or unstable depends on the relative strengths or rates of potential interactions. This is often manifested as instability when interactions between system components are dominant, and stability when self-limiting (or externally limited) properties of system components are dominant. Two simple examples are given below to illustrate this.

4.1 Thickness of forest soils

Returning to the example discussed above, the relationships among tree establishment, soil thickness, and site favorability are shown in Fig. 3.2. The model also shows self-limits on tree establishment, due to factors other than soil quality and thickness that constrain tree establishment, and on root penetration of bedrock, due to the limits of the root zone for a specific tree species.

Methods for evaluating the stability of systems as shown in Fig. 3.2 are described by, e.g., Puccia and Levins (1985), Logofet (1993), and Phillips (1999). In this case the system is conditionally stable, contingent on the relative strength of:

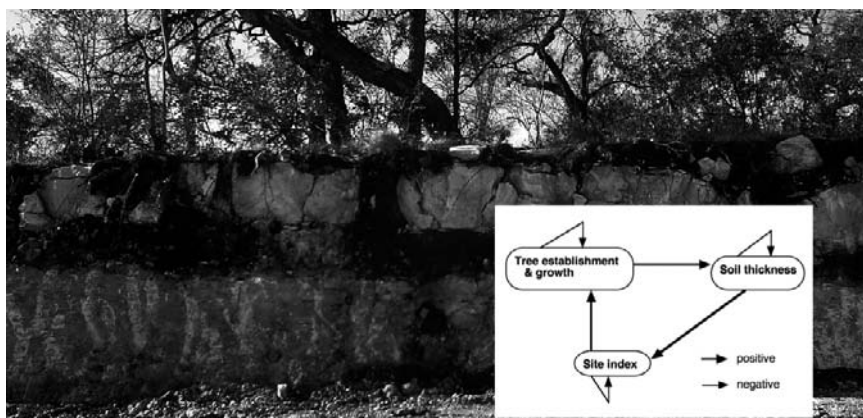


Figure 3.2 Conceptual model of interactions among tree establishment, soil thickness, and site index, superimposed on photo of local variation in soil thickness associated with repeated tree exploitation of bedrock joints (Comal County, Texas; outcrop is about 1.7 m high).

- The loop whereby tree establishment leads to greater root penetration of bedrock, which increases local soil thickness, with positive feedback to tree establishment; and
- Self-limiting effects on tree establishment and root penetration.

Where the former is stronger, divergent evolution of soil thickness occurs, and pedodiversity (at least with respect to soil depth/thickness) increases. However, as soil thickness becomes less of a limiting factor for tree success, and depth approaches typical rooting depths, the self-limitations become dominant and soil thickness as influenced by effects of individual trees would converge toward the typical rooting depth. Thus, while soil development might well undergo an extended period of divergence and increasing pedodiversity, eventually a pedogenetic mode switch would occur, limiting the divergence.

4.2 Chemical weathering

For rock of a given composition, and within a given climate setting, the first order control on the rate of chemical weathering is the amount of moisture the rock is exposed to. Faster or slower weathering rates produce greater or lesser degrees of weathering. The degree of weathering, in turn, influences moisture penetration via increases in porosity and hydraulic conductivity. The degree of weathering has a negative influence on weathering rates, due to depletion of weatherable minerals. These interactions are shown in Fig. 3.3, along with the self-limiting effects on moisture and weathering rates. The latter are associated with hydroclimatological limits on

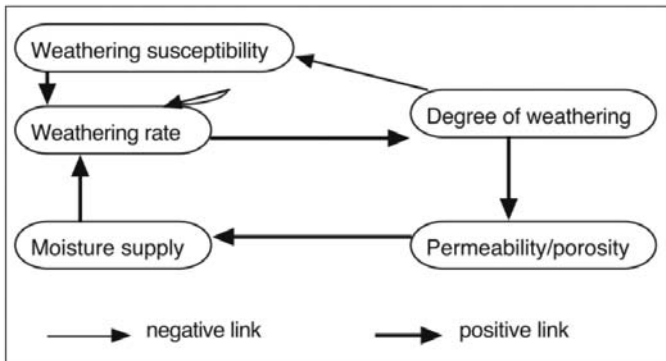


Figure 3.3 Interactions in chemical weathering in soils and weathering profiles.

moisture supply, and chemical kinetic limits on weathering rates. Phillips (2001a) examined these phenomena in a study of inherited vs. acquired characteristics of weathering profiles.

The system is conditionally stable, again dependent on the relative strength of self-limiting effects vs. interactions between system components (moisture effects on weathering rates; weathering rate effects on degree of weathering; feedbacks to moisture input). Differentiation in weathering status increases on average when instability occurs when mutual interactions among system components is dominant. When moisture or weathering rate limits come into play, the system is stable and weathering is convergent.

5. Lessons from Lexington

Much has been written in this volume and elsewhere about the effects of spatial scale or resolution on (assessments of) soil richness, evenness, differentiation, and spatial variability. A related but distinct issue, however, are the effects of the intensity of inventory and mapping of both soils and SFF. Intensity is used here as a shorthand for cumulative levels of effort and detail put into the survey of soils and SFF. This can be illustrated via an example.

5.1 Soil and parent material variations of the Lexington Limestone

The upland interfluvial areas (that is, those areas not within stream valleys or along valley side slopes of incised valleys) of the Inner Bluegrass region of central Kentucky, USA, are underlain by an upper Ordovician geologic formation known as the Lexington Limestone. Published soil surveys, the U.S. Department of Agriculture's (USDA) Web Soil Survey maps (<http://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx>), and the USDA

Official Series Descriptions database (<http://soils.usda.gov/technical/classification/osd/index.html>), which includes lists of geographically associated soils for each series, were used to identify soil series mapped (not always exclusively) in areas of the Inner Bluegrass where the surficial geology is the Lexington Limestone formation. The latter areas were identified based on 1:24,000 scale geologic mapping obtained via the Kentucky Geological Survey.

Fifteen soil series are mapped on the Lexington Limestone. How does this compare with parent material variability?

Clepper (2011) examined the lithostratigraphy of the Lexington Limestone in central Bluegrass in great detail. Her review of previous work shows that as of the 1970s, 14 different members or tongues (discontinuous lenses or layers of sedimentary rock) were recognized. By the late 20th century, stratigraphic research and field mapping had raised that number to 20. Clepper's own results, based on 290 exposures or outcrops of various kinds, identified 28 different facies or lithostratigraphic units (Clepper 2011, Table 2.1).

It would, therefore, appear that soil richness is less than the richness of the parent material, suggesting convergent pedogenesis. However, this might be misleading. Soils of the Inner Bluegrass region have been mapped only once in any given area since the current U.S. Soil Taxonomy, or its immediate and related predecessors, came into use, and no county soil survey in the area was published more recently than 1983. Of the 15 series, the USDA database shows that 10 were established between 1915 and 1943 (Culleoka, Donerail, Eden, Fairmount, Loradale Lowell, Maury, McAfee, Nicholson, and Salvisa series). Three series (Caleast, Cynthiana, Faywood) were established in the 1960s, one in 1979 (Chenault), and only one within the past five years (Bluegrass series 2008). Therefore the parent material has been studied and inventoried much more intensively than the overlying soils, at least with respect to soil morphology and stratigraphy.

Further, series descriptions generally mention parent geology in only a very general sense (e.g., limestone, phosphatic limestone, limestone with interbedded shale). The distinctions among the 15 series are mainly with respect to drainage class, presence of mollic features or a mollic epipedon, presence of a loess cover over the limestone residuum, and total soil thickness. Some of these are indirectly related to the underlying geology—for instance, rock composition may influence cation exchange capacity and pH of the overlying soil, and thus whether mollic features are identified; or the complex subsurface karst flow network may influence drainage class. The extent to which variations associated with the 28 facies identified by Clepper (2011) are associated with significant variations in the overlying soils is simply unknown.

5.2 Local and regional pedodiversity

The same study area can be used to illustrate the phenomenon of profoundly different implications with respect to pedodiversity and soil variability at regional and local scales.

While the regional inventory of mapped soil taxa compared to parent material variability does not suggest divergent evolution, detailed local-scale analyses may turn up multiple soils on a single parent material. For example, Fig. 3.4 shows soil and weathering profiles temporarily revealed by excavation in Madison County, Kentucky in 2010. At least five morphologically distinct soils are evident over a 5 m distance in a single lithotype of the Lexington limestone. In this case the variations appear to be largely related to instabilities associated with weathering feedbacks in the underlying parent material.

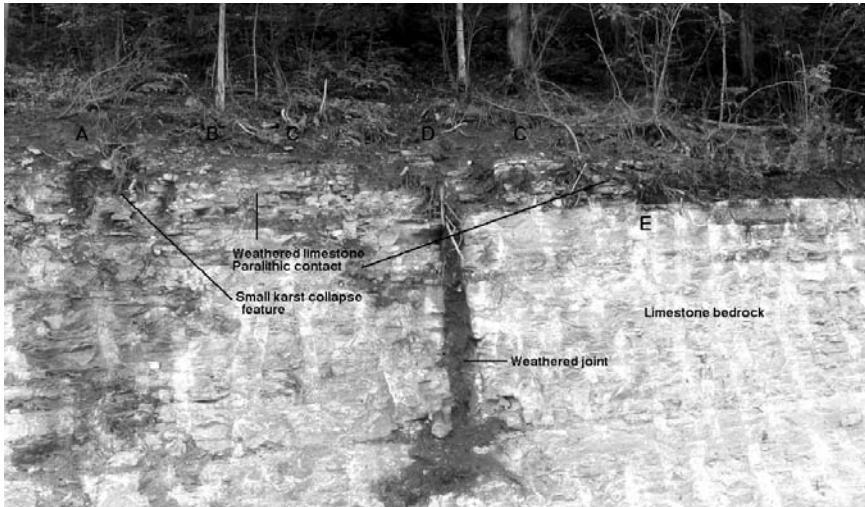


Figure 3.4 Soils exposed in an outcrop of a single lithostratigraphic unit of the Lexington Limestone Formation in Madison County, KY. Exposure is about 5 m long and 2 m high. Large letters mark morphologically distinct soils. A) dark, possibly mollic surface horizon overlying a Bt horizon, and a thick transition zone of rock fragments and a Cr horizon of weathered limestone. B) Similar to A, but with a much thinner zone of weathered rock. C) similar to B, but lacking the dark surface layer. D) Thick soil with dark surface layer, multiple argillic horizons, and little or no Cr layer in transition to bedrock. E) Thin, dark, possibly Mollic soil with abrupt transition to bedrock.

Conclusions

Complex nonlinear dynamics in pedogenesis, and in the interactions among soils, landforms, biota and climate, often results in dynamical instabilities and deterministic chaos. This means that the effects of small disturbances, or of variations in initial conditions, can persist and grow over time, leading to divergent pedogenesis. Where divergent pedogenesis occurs, it results in increasing pedodiversity over time, and to soil variations that are large relative to those of soil forming factors.

Neither divergent or convergent pedogenesis can continue indefinitely and the appearance of either can vary not only with the environmental situation, but also with spatial or temporal scale or intensity of survey. Two phenomena are common in this regard. One is a tendency toward instability and divergence when interactions among Earth surface system components dominate system dynamics, which is often more common in earlier stages of development. As external controls or self-limitations on development of system components come into play, stable, convergent development ensues. The second is a tendency for convergence at a regional (soil landscape or broader) scale, as the range of variation of soil forming factors constrains pedodiversity to a finite set of outcomes. This is coupled with a tendency for divergence to be noted at a more local scale, where pedological variation within the set of possible outcomes is often disproportionately large relative to that of soil forming factors.

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CHAPTER 4

Fractals and Multifractals in Pedodiversity and Biodiversity Analyses

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Caniego Monreal^b

Why Fractals and Multifractals are Needed to Understand Pedodiversity and Biodiversity

Origin of Fractals and Multifractals

Since the publication of B. B. Mandelbrot's book "The Fractal Geometry of Nature" (1977), much has been written about the use of fractal geometry and multifractal analysis as new mathematical tools to better understand natural shapes, processes and systems. A complementary approach to this question can be found in the book by I. Stewart, "Does God Play Dice?" (1989). Under this eye-catching title Stewart introduces the state of the art of the mathematics that links fractals and multifractals with two major theories of mathematical physics, the theory of dynamical systems and the theory of chaos and nonlinear dynamics. The origins of

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the mathematical discipline of dynamical systems can be traced back to Isaac Newton in the second half of the 17th century, while the theory of chaos and nonlinear dynamics was initiated by Henri Poincaré at the end of the 19th century. It is perhaps a coincidence that Newton developed his theory to explain planetary motions, while Poincaré extended it by introducing new geometrical methods to determine the stability of the solar system. Poincaré investigated this question in the reduced setting of the asymptotic dynamics of three bodies moving under the mutual gravitational attraction of Newton's second law. Nowadays the theory of chaos and nonlinear dynamics is a fundamental in understanding phenomena in physics, chemistry, biology and geosciences. In this context, fractals and multifractals provide mathematical tools to characterize the dynamics of chaotic nonlinear dynamical systems. This framework was introduced by soil scientists (Smeck et al. 1983, Culling 1987, 1988, Ibáñez et al. 1990, Ibáñez and García-Álvarez 1991, Phillips 1993, 1998, 1999, 2001) for Earth surface systems from Jenny's version of the state-factor model for soil formation (Jenny 1941). This view was initiated in the late 19th century. At that time the Russian geologist, geographer and soil scientist, V.V. Dokuchaev, argued that soils are a unique object produced by the combined action of animals, plants, climate, relief, geology and time.

Geometric irregularity is an intrinsic property of Earth-surface systems. Soil pores and particles, organisms living in soils, vegetation, landforms and river networks have a wide range of sizes and shapes. Soil and crop canopy surfaces are rough, and patterns on soils, yield maps, landforms and fluvial systems have complex shapes. This geometric irregularity is easy to perceive and observe, but quantifying it has long presented a daunting challenge. Such quantification is imperative because geometric irregularity is both the cause and reflection of spatial and temporal variability that in turn strongly affects soil and crop management, and, in general, any Earth environmental system.

Power laws, a fingerprint of fractal geometry (Mandelbrot 1977), have been observed in the context of ecology and other natural sciences, and fractals have been applied to model spatial and temporal patterns (Sugihara and May 1990, Hastings and Sugihara 1993). In particular, species-area relationships, which describe the number of species in a given area, conform to a power law (May 1975, 1981, Gaston 1994, and Rosenzweig 1995). This power law model of the richness-area relationship has been reinterpreted in terms of the "self-similarity" of the spatial distribution of species abundance (Harte et al. 1999). It reflects the existence of invariance in heterogeneity over a range of scales.¹ Moreover, Harte et al. (1999) set up a fractal model

¹Scales or measurement scales are understood as lengths, areas or volumes within which the shape irregularity is ignored and curves or surfaces are assumed to have a simple geometric shape, i.e., line segment or part of a plane.

based on this self-similar hypothesis that provided a power law for the corresponding richness-area relationship.

In the context of pedodiversity, the pedorichness-area relationships conform to the well-established pattern in ecology, the power-law model, $T(A)=cA^z$, where $T(A)$ is the number of different soil taxonomic entities within an area A ; c and z are constants (Ibáñez et al. 2005). Moreover, the exponent z coincides with observations reported in the ecological literature (Ibáñez et al. 2005). The power-scaling behavior of the pedorichness-area relations may also be interpreted in terms of the self-similarity of the pedorichness spatial distribution. Thus, the power law becomes the by-product of a fractal structure while the scaling exponent quantifies the regularity trend in the reproduction of the spatial pattern of pedorichness within a certain range of scales. This fractal geometry of pedorichness spatial distribution patterns has been investigated (Ibáñez et al. 2009). The patches of each pedotaxa in Europe were considered as a subset of points in two-dimensional space. When these flat objects exhibit complex self-similar geometric features (exact or statistical), a natural way to describe them is through their fractal dimension. Self-similar sets are those whose parts are (exactly or statistically) similar to the whole geometrical object over a range of spatial scales. This self-similarity gives rise to scale invariance which in turn provides, as a by-product, a power law whose exponent becomes the fractal dimension of the set. Fractal shapes in a two-dimensional plane typically have a fractional dimension between 1 and 2. This dimension gives a measure of the “geometrical size” of the object, and it is related to the way it fills up the plane (Falconer 1990). The patches of the 132 European pedotaxa considered by Ibáñez et al. (2009) displayed a well-developed fractal behavior with fractal dimensions ranging between 1.07 and 1.6.

Pedotaxa-abundance distributions, as mathematical objects, may be conceptualized as probability measures or mass distributions. What is the nature of these measures? As stated before, the state of soil or the state of a particular soil property may be represented as the outcome of a chaotic nonlinear dynamical system and highly irregular patterns, with multifractal behavior should be common (Beck and Schlögl 1995). If we assume the self-similar hypothesis of Harte et al. (1999) for soils, pedotaxa-abundance distributions should exhibit an extremely complex but self-similar behavior. A mathematical model for self-similar measures that exhibit multifractality was proposed by Hutchinson (1981). This model is based on the so called multinomial measures. In this context it has been suggested that pedotaxa-abundance distributions should display a complex behavior similar to multifractal self-similar patterns (Caniego et al. 2006) close to multinomial measures.

Multifractal scaling was first introduced by Mandelbrot (1974) in the context of “turbulence,” but the term was coined later. In order to characterize fractal objects and strange sets of nonlinear physics, multifractal theory was elaborated under the ideas of thermodynamic formalism (Frisch and Parisi 1985, Halsey et al. 1986). Multifractal structures have been described for very different fields such as diffusion-limited aggregation (DLA) patterns (Mandelbrot and Evertsz 1991), earthquake distribution analysis (Hirata and Imoto 1991), signal processing (Lévy Véhel and Mignot 1994), internet data traffic modeling (Lévy Véhel and Riedi 1997), characterization of mineral deposits (Cheng et al. 1994), analysis of vegetation patterns (Scheuring and Riedi 1994, Borda-de-Agua et al. 2002), zooplankton biomass time series analysis (Pascual et al. 1995), characterization of soil porosity and particle size distributions (Caniego et al. 2001, Martín and Montero 2002), soil spatial variability (Kravchenko et al. 1999, Caniego et al. 2005, San José Martínez et al. 2010), and soil microtopography (San José Martínez et al. 2009).

As fractal objects are geometrical self-similar sets (in an exact or statistical sense), multifractal objects are measures, which exhibit self-similarity (exact or statistical). A measure is simply a way to specify a rule of spreading a mass or probability over a set that is called the support of the measure. Pore-size distributions of soil or pedotaxa-abundance distributions, among others, are examples of this. The simplest model of such a multifractal measure is the binomial measure that is constructed as follows (Hutchinson 1981). The unit mass is spread over each half of the unit interval (the support of the measure) with non-equal relative proportions. And that way, this rule is repeated again for each half of the unit interval. When this process is iterated, it yields a strictly self-similar measure that has no density; it is a singular measure. But more important, it shows “burst” and “jumps”, and erratic and highly irregular behavior. Multifractal analysis characterizes this complex pattern by the scaling exponent associated with the singularity of each point of the support of the measure (Cawly and Mauldin 1992 and Evertsz and Mandelbrot 1992). Typically, singularity exponents of multifractal distributions fill up a whole interval, and the support of the measure is split into a plethora of interwoven sets of fractal nature, each one containing the points with the same singularity exponent. The singularity spectrum is the function that assigns to each singularity exponent the fractal dimension of the set of points with this exponent. Thus, the complexity of the measure is characterized by a whole set of singularity exponents and their fractal dimensions.

A complementary approach to multifractal analysis that comes from the information theory is based on Rényi entropies (Rényi 1970). The spectrum of generalized dimensions or Rényi spectrum (Grassberger 1985) generalizes the notion of entropy dimension by introducing entropies of any order. This spectrum summarizes a number of commonly used diversity

indices through the exponent of their scaling behaviors (Borda de Agua et al. 2002). In this way multifractal analysis provides a unified framework to investigate pedodiversity, besides pedorichness-area relationships and pedotaxa-abundance distributions (Caniego et al. 2007).

1. The Mathematics Behind Fractals and Multifractals

1.1 Fractals

Geometrical models of fractal objects can be generated iteratively, by repetition of the same form- or shape-changing operation at different scales. An example of such a construction is the Koch curve (Fig. 4.1). The initial shape, or initiator, is a segment. Then, the initiator is divided in three equal parts, replacing the middle part by an equilateral triangle and taking away its base. This is the shape-changing operation, or generator. To start up the iterative process, four copies of the generator are scaled down by a factor of 1/3 to replace the four segments of the generator. Thus, we now repeat the shape changing operation, replacing each resulting line segment by a reduced copy of the generator. The Koch curve is self-similar because it can be broken down into arbitrary small pieces, each of which is a small replica of the entire structure. If the Koch curve is reduced by a factor of one third and four copies are made, then these copies can be pasted together to give back the Koch curve; one ninth reduction requires 16 copies, one twenty seventh reduction requires 54 copies, etc. In all self-similar objects, there is a relationship between the reduction factor R and the number of pieces N into which the object is divided. This relationship follows a power-law

$$N=R^{-D_{ss}} \tag{1}$$

where D_{ss} is called the self-similarity dimension. The value of D_{ss} can be found by equating the logarithms of both sides of Eq. (1), i.e., $D_{ss} = -\log(N)/\log(R)$. For the Koch curve, the reduction factor is $(1/3)^k$

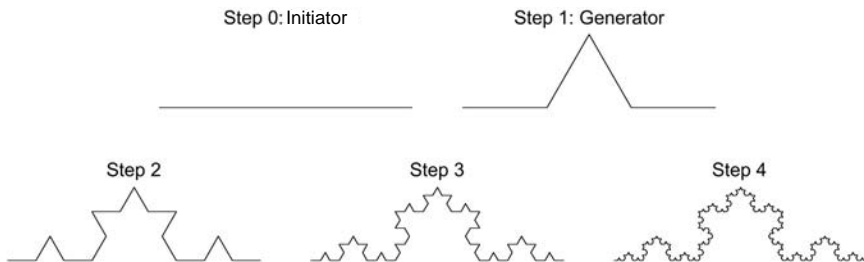


Figure 4.1 Koch curve.

at step k and 4^k reduced copies are needed to recover it. Then the self-similarity dimension is 1.261.

$$D_{ss} = -\log(4^k) / \log(1/3^k) = \log(4) / \log(3) = 1.261$$

The Sierpinski carpet is another example of a self-similar fractal (Fig. 4.2). The square is the initiator in this case. The generator is obtained by dividing the square into nine smaller equal squares and extracting the one from the center. If the Sierpinski carpet is reduced by a factor of one third and eight copies are made, then these copies can be pasted together to give back the Sierpinski carpet; one ninth reduction requires 64 copies, one twenty seventh reduction requires 512 copies, etc. Then, for a reduction factor of $(1/3)^k$ at step k , 8^k reduced copies are required to give back the Sierpinski carpet. Then the self-similarity dimension is 1.892, i.e.,

$$D_{ss} = -\log(8^k) / \log(1/3^k) = \log(8) / \log(3) = 1.892$$

It is worth noting that if the self-similarity dimension of regular geometrical objects, like segments, squares or cubes were computed we would get the dimension of the space where these objects reside, one for segments, two for squares and three for cubes. Fractals sets are intermediate geometrical objects. The Koch curve has infinite length as a one-dimensional point set and the Sierpinski carpet has no area as a two-dimensional set. In

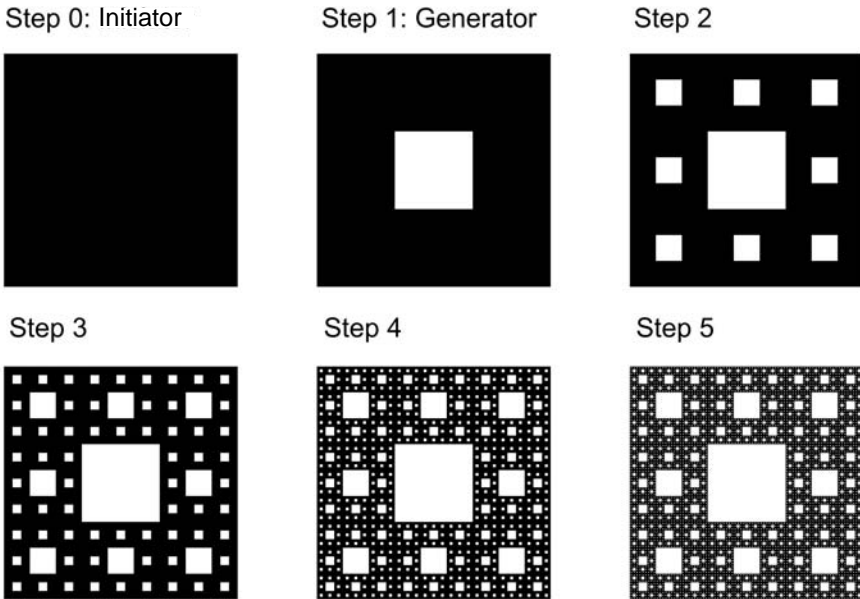


Figure 4.2 Sierpinski carpet.

both cases the power law is verified for non-integer numbers as opposed to regular sets such as segments or squares. Felix Hausdorff, the German mathematician, who before World War II, worked out a notion of dimension so that non-regular sets of points like the Koch curve or the Sierpinski carpet would have a finite length and non-zero area when computed with the right dimension. Hausdorff's notion of dimension corresponds to the self-similarity dimension for self-similarity point sets. This was the motivation for the notion of fractal dimension introduced by Mandelbrot (1977).

Ideal shapes like those depicted in Figs. 4.1 and 4.2 can be used as rough approximations of real objects in the same way as geometric lines, spheres, cylinders, etc. are used. Applying random changes in the generator procedure result in the construction of stochastic fractals that mimic natural shapes (Peitgen et al. 1992). Because random changes are applied, realizations of stochastic fractals with the same initiator and generator differ. Figures 4.3 and 4.4 show a random Koch curve and a random Sierpinski carpet. They present more realistic models for coastlines or two dimensional sections of a porous material. Stochastic fractals retain the important feature of dependence of geometric properties on scale, but this feature is preserved in a statistical sense. This means that Eq. 1 will not hold exactly for any realization of stochastic fractals at any scale. Nevertheless, the dependence of the property of interest on scale for each realization is close to the same power law dependence.

One approach when characterizing pedodiversity is to examine the geometrical structure of the spatial distribution of soil pedotaxa. In doing this we consider the areal extent of each pedotaxa and regard this as a subset of points on the two-dimensional plane (Ibáñez et al. 2009). Here we are following a standard procedure similar to the one used by plant ecologists when analyzing biodiversity of plant communities in which they consider the areal extent of each species (Magurán 1988, Scheuring and Riedi 1994). Then, the complex geometrical structure of the spatial distribution of soil types may be assessed through its fractal dimension as estimated by the box-counting dimension. To estimate the box-counting dimension of the

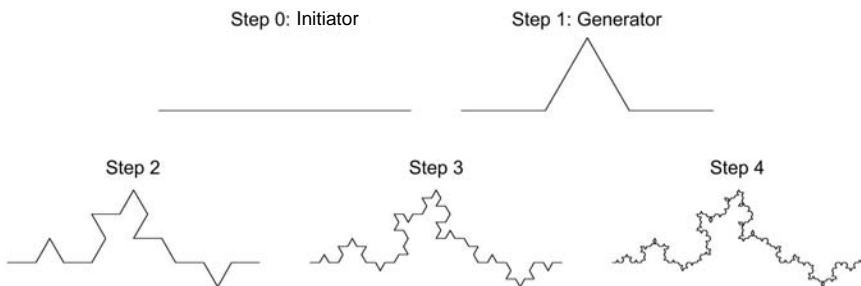


Figure 4.3 Random Koch curve.

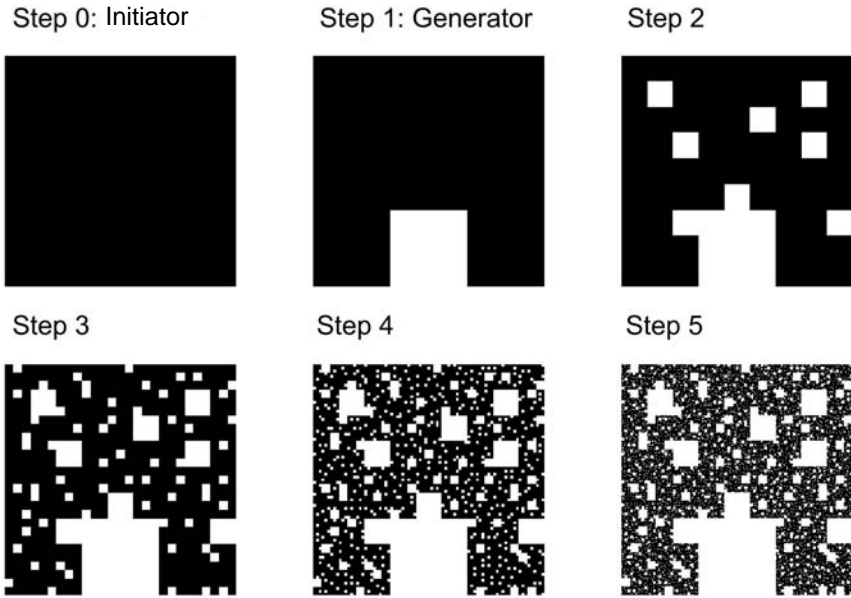


Figure 4.4 Random Sierpinski carpet.

shape of the area occupied by a specific pedotaxa, let us consider a mesh of grid size ε and count the number of cells $N(\varepsilon)$ of the mesh that are needed to cover the shape at the scale ε . Typically, for a self-similar two-dimensional set these numbers follow a power law rule:

$$N(\varepsilon) \propto \varepsilon^{-D}$$

Here, symbol \propto stands for asymptotic behavior as ε approaches zero. Then, the fractal dimension D is estimated by linear regression of the pairs $(\varepsilon, N(\varepsilon))$ in a log-log plot. For real shapes this power law relationship holds for a finite range of values of ε or scales. Then one has minimum and maximum observable scales, or lower and upper cutoff scales beyond which this power law relationship does not hold.

1.2 Self-affine fractals

Data on surface roughness or topography along spatial transects do not follow the self-similarity law of Eq. 1, but demonstrate another, anisotropic type of scale-dependence, called self-affinity. It means that the elevation (height) $Z(\lambda x, \lambda y)$ corresponding to a point $(\lambda x, \lambda y)$ in two-dimensional space is statistically similar to $\lambda^H Z(x, y)$, where $0 \leq H \leq 1$ is called the Hurst exponent. That is, changing the horizontal measurement scale λ times has to correspond to changing the vertical measurement scale λ^H times. While

self-similarity may be considered an isotropic-scaling law, which would mean that in two-dimensional space, $Z(\lambda x, \lambda y)$ would be statistically similar to $\lambda Z(x, y)$ where λ is a reduction factor, self-affine fractals are anisotropic, which means that in two-dimensional space, $Z(\lambda x, \lambda y)$ is statistically similar to $\lambda^H Z(x, y)$. Self-affine surfaces are observed in three dimensions, so that the vertical scale has to be decreased λ^H times and scales in horizontal directions have to be decreased λ times to obtain a statistically similar surface. Random algorithms based in this statistical property have been used to generate quite realistic landscapes (see Feder 1988).

Statistical laws of the kind $P(\lambda x) \approx \lambda^H x$ define a type of random variation known as fractal Brownian noises that were introduced by Mandelbrot and Van Ness (1968). In the geosciences $P(x)$ may be thought of, for instance, as the value of a soil measurable property of a point at a distance x in a catena or the value of the property at a given site in the instant x . The exponent H is called the Hurst exponent. High values of the Hurst exponent indicate memory or autocorrelation in the data. Low values suggest an anti-correlation or self-correcting response. Scaling analysis of observed measurements at different points (or instants) may be implemented in order to determine the structure of the variation and quantify the degree of its memory. Fractal noises have been used as fractal models to explain soil spatial variability first by Burrough (1981, 1983). Temporal variability was also examined by Eghball and Powell (1995) and Eghball et al. (1999). Later, these models, and the multifractal variant, were widely used to describe random variations in soil science (Kravchenko et al. 1999, Kravchenko and Pachepsky 2004, Eghball et al. 2003, Caniego et al. 2005, Zeleke and Si 2006, San José Martínez et al. 2009, 2010).

1.3 Multifractals

Multifractals present yet another example of self-similarity that has been successfully applied to simulate and parameterize properties of soils and other natural systems. The irregularity now is related with the heterogeneity of distributions on a set of points rather than to geometric objects. Soil, considered as a porous media, is highly complex in regard to the arrangement of the solid and void phases, as well as to the distribution of their constituent materials or organic matter content within the soil matrix. Measurements of some of soil property (such as the amount of a given component or the void space), assigns to each (1-D, 2-D or 3-D) region R the corresponding quantity $\mu(R)$. Typically, $\mu(R)$ depends on the location of the particular region in the medium and varies widely in relation to the volume of R . A model of a multifractal measure (see Fig. 4.5) appears as the result of a multiplicative cascade starting (step 0) with an initial amount $\mu(S)$ corresponding to the support S (e.g., organic matter

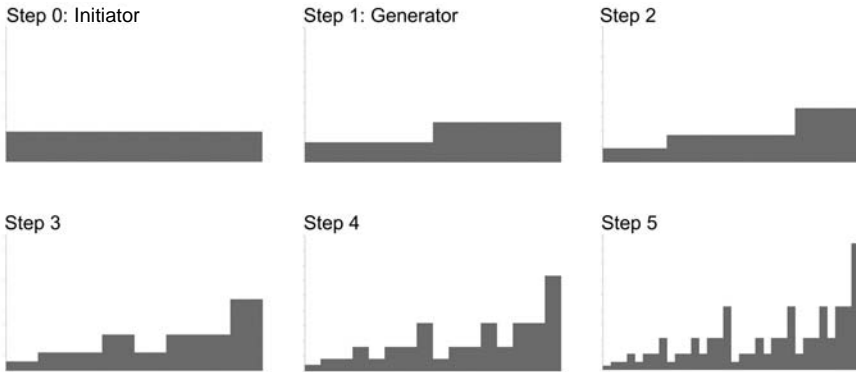


Figure 4.5 The binomial multifractal measure with weights 1/3 and 2/3.

content along transects). In the following scale step (say, by dividing the support S in two equal sub-interval S_i) each quantity $\mu(S_i)$ corresponding to the measure of S_i is determined by a multiplicative factor $w_i < 1$, that is, $\mu(S_i) = w_i \mu(S)$, $\sum \mu(S_i) = \mu(S)$, continuing this process at *infinitum*. The above process leads to an infinite range of concentrations. Multifractal analysis helps to organize heterogeneity by representing it as a superposition of homogeneities of different degrees of intensity summarized by means of the so called multifractal spectrum $f(a)$, which reports of the size (fractal dimension) of the sets of points of concentration degree a (this being measured in adequate terms by the Hölder exponents). Let us now discuss how to estimate multifractal parameters.

For the sake of clarity let us consider a specific problem: the characterization of pedotaxa abundance distribution in a particular region. Consider the one-dimensional distribution of relative fraction of area occupied by a pedotaxa in that region (Caniego et al. 2006). Therefore, it is a one dimensional distribution over the unit interval. To investigate this distribution let us generate a partition of size ε that covers the interval of sizes (fractional areal extent) of the pedotaxa present in the region of interest. Then, let us call $I_i(\varepsilon)$ the i -subinterval ($i = 1, 2, \dots, n(\varepsilon)$) generated by the partition of size scale ε , being $n(\varepsilon)$ the number of subintervals of this partition. The quantity $\mu_i(\varepsilon) = \mu(I_i(\varepsilon))$ is the measure or mass proportion of $I_i(\varepsilon)$: then $\mu_i(\varepsilon)$ corresponds to the total fraction of pedotaxa with sizes belonging to $I_i(\varepsilon)$. In order to assess the multifractal patterns of measure μ , the scaling exponents $\alpha_i(\varepsilon)$,

$$\mu_i(\varepsilon) \propto \varepsilon^{\alpha_i(\varepsilon)}$$

for the quantities $\mu_i(\varepsilon)$ are considered. For most cases the exponent equals one and the measure has a density function. This is what happens with a standard statistical distribution like the normal distribution. Then,

the ratio $\mu_i(\varepsilon)/\varepsilon$ is the empirical density of the cell $I_i(\varepsilon)$ and it defines the density function as ε approaches zero and the ratio converges to a non-zero value. In other cases $\mu_i(\varepsilon)/\varepsilon$ diverges or tends towards zero and it has no density function. In these cases, the asymptotic behavior of $\mu_i(\varepsilon)/\varepsilon^{\alpha_i(\varepsilon)}$ is well defined when a suitable exponent $\alpha_i(\varepsilon)$ is chosen but, in general, this exponent varies from one cell to another and, eventually, from one point to another as ε approaches zero (Evertsz and Mandelbrot 1992). These are the singular measures that exhibit a variety of scale dependences that multifractal analysis aims at characterizing through the singularity or Hölder exponents α which correspond to the asymptotic behavior of the coarse exponents $\alpha_i(\varepsilon)$. Typically, singularity exponents α of multifractal distributions show a great variability such that their values fill an interval $[\alpha_{\min}, \alpha_{\max}]$. When this interval reduces to a point the measure becomes a (mono-)fractal.

Multifractal analysis relies on the idea of grouping together cells $I_i(\varepsilon)$ of similar degrees of mass concentration α and analyzing the growth rate of the number of cells $I_i(\varepsilon)$ as the size ε approaches to zero. Specifically, if $N_\varepsilon(\alpha)$ is the number of subintervals of size ε with singularity exponent between α and $\alpha + \Delta\alpha$, and $f(\alpha)$ is defined by the scaling relation:

$$N_\varepsilon(\alpha) \propto \varepsilon^{-f(\alpha)}$$

then, as $\Delta\alpha \rightarrow 0$, $f(\alpha)$ converges to a continuous function called the multifractal spectrum (Evertsz and Mandelbrot 1992). Typically, it has a concave down-bell shape (Fig. 4.6). Following Chhabra and Jensen (1989), the singularity spectrum may be computed using a set of real numbers q by

$$\alpha_q \propto \frac{\sum_{i=1}^{m(\varepsilon)} \mu_i(q, \varepsilon) \log \mu_i(\varepsilon)}{\log \varepsilon} \tag{2}$$

and

$$f(\alpha_q) \propto \frac{\sum_{i=1}^{m(\varepsilon)} \mu_i(q, \varepsilon) \log \mu_i(q, \varepsilon)}{\log \varepsilon} \tag{3}$$

where the quantities $\mu_i(q, \varepsilon)$ are defined as

$$\mu_i(q, \varepsilon) = \frac{\mu_i(\varepsilon)^q}{\sum_{i=1}^{m(\varepsilon)} \mu_i(\varepsilon)^q}$$

The summation is taken over the $m(\varepsilon)$ cells with no zero mass. In general this spectrum has a parabolic shaped concave-down graph that attains its

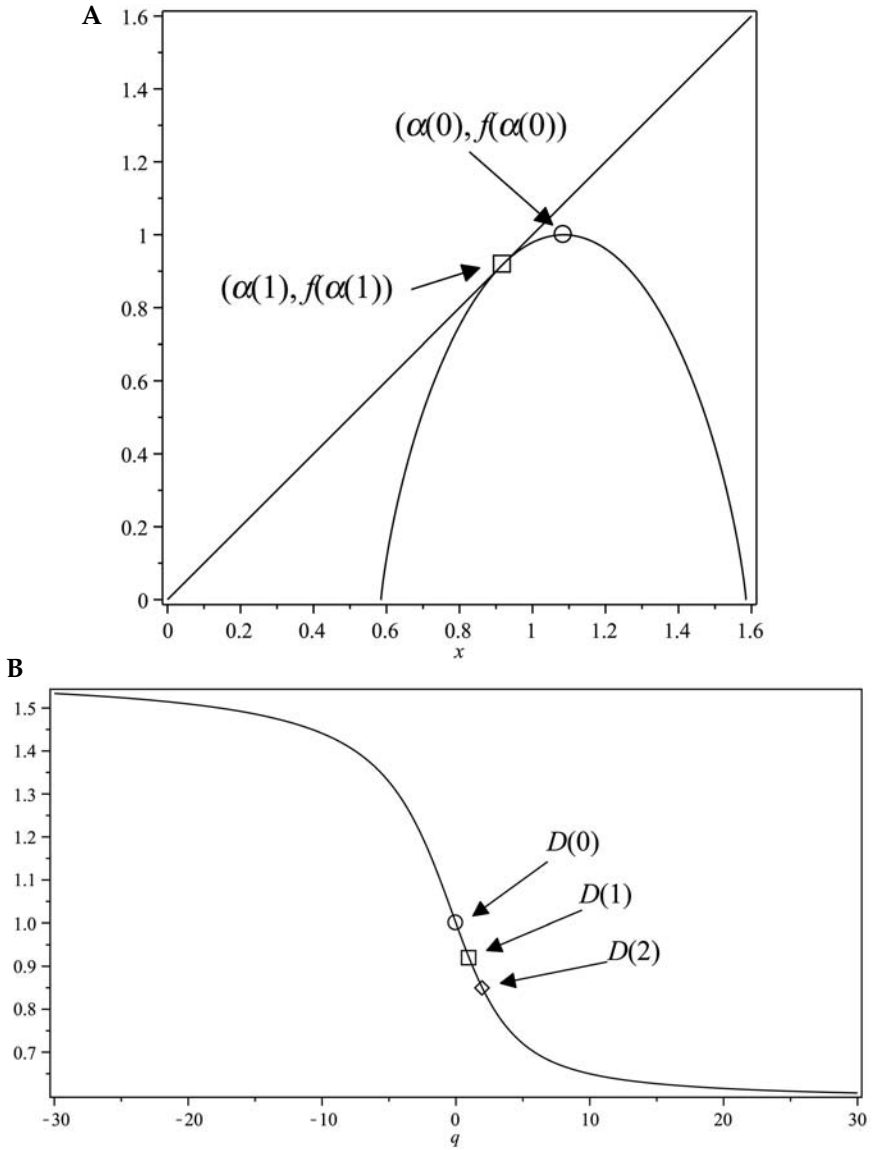


Figure 4.6 Multifractal (A) and Rényi (B) spectra of the binomial measure with weights 1/3 and 2/3.

maximum value $f(\alpha_0)$ at the point α_0 . The singularity exponent α_0 corresponds to the mean value of the singularity exponents α as it is the average of the exponents α weighted by the uniform distribution (Eq. 2), and $f(\alpha_0)$ gives the box-counting fractal dimension of the support of the measure (Eq. 3). This will be one for a one-dimensional transect and two for a two-dimensional plane. In general it will be the fractal dimension of the support when this is a fractal itself.

Another way to compute the singularity of multifractal measures comes from information theory through the generalization of Shannon entropy. Rényi (1961, 1970) introduced the Rényi or generalized dimensions D_q

$$D_q \propto \frac{1}{q-1} \frac{\log \sum_{i=1}^{m(\varepsilon)} \mu_i(\varepsilon)^q}{\log \varepsilon} \tag{4}$$

for $q \neq 1$ and

$$D_1 \propto \frac{\sum_{i=1}^{m(\varepsilon)} \mu_i(\varepsilon) \log \mu_i(\varepsilon)}{\log \varepsilon} \tag{5}$$

for $q = 1$. It turns out that the dimension $D_1 = f(\alpha(1)) = \alpha(1)$ corresponds to the entropy dimension of the distribution and $D_0 = f(\alpha(0))$ is the box-counting dimension of the geometrical support of the measure, both ranging between 0 and 1 for one-dimensional distributions. While the singularity spectrum is a concave function, the Rényi spectrum D_q is a sigma-shaped, decreasing function with respect to q (Fig. 4.6).

1.4 Multifractal diversity indices

A number of indices have been introduced in the literature devoted to the study of diversity. Margalef (1958) was the first to use Information Theory to study ecological diversity in the search for general laws that explain ecosystem structures. Rényi information theory (Rényi, 1961) summarized a number of diversity indices in the following way. If P_i ($i = 1 \dots N$) are the probabilities of a discrete random variable with possible outcomes x_i obtained for example by coarse-graining a continuous variable, the Rényi information functions are defined as $H_q = [1/(1-q)] \sum_{i=1}^N P_i^q$ ($q \neq 1$), and $H = -\sum_{i=1}^N P_i \log P_i$. Let us note that $\lim_{q \rightarrow 1} H_q = H$ is the Shannon entropy. Now let us consider that p_i is the areal proportional contribution of each soil type to total sampled area with N different

soil types. In this case Shannon entropy has been found to be a useful index of diversity. Other indices that are usually considered are richness, $R = N$, Simpson index, $S = \sum_{i=1}^r p_i^2$, evenness, $E = H / H_{\max}$ (H_{\max} being the maximal entropy) and Berger-Parker index, $B = \max_i \{p_i\}$ (Magurran 1988, Ibáñez et al. 1998, Borda-de-Agua et al. 2002).

Now, let us consider pedotaxa-abundance distributions as a probability measure or mass distribution and let $\mu_i(\varepsilon)$ be the measure or mass of the i -cell of the partition with cell size equal to ε , then $\sum_{i=1}^{n(\varepsilon)} \mu_i(\varepsilon) = 1$, where $n(\varepsilon)$ is the number of cells of size ε . For a given partition of size ε and mass proportion $\mu_i(\varepsilon)$ in each cell, one has the entropy or Shannon index,

$$H(\varepsilon) = - \sum_{i=1}^{m(\varepsilon)} \mu_i(\varepsilon) \log \mu_i(\varepsilon)$$

where the summation is taken over the cells with non-zero mass. Then, the entropy dimension D_1 (Eq. 5) gives the growth rate of the Shannon index as the size of the partition shrinks. When an adequate scaling behavior takes place for an experimental measure, D_1 provides a physical characterization by measuring how its heterogeneity evolves across a certain range of scales. The correlation or Simpson index is defined as (Scheuring and Riedi 1994, Borda-de-Agua et al. 2002).

$$S(\varepsilon) = C_2(\varepsilon) = \sum_{i=1}^{n(\varepsilon)} \mu_i(\varepsilon)^2$$

For multifractal measures the growth of the Simpson index with respect to the size partition follows a power law being the correlation dimension D_2 , the exponent of this scaling (Eq. 4 for $q = 2$). Similarly, let us consider higher-order correlations,

$$C_q(\varepsilon) = \sum_{i=1}^{n(\varepsilon)} \mu_i(\varepsilon)^q$$

then, D_q generalizes the correlation dimension for arbitrary real value of q . The scaling behavior of the Simpson index and the other correlation dimensions over a range of scales, as well as the fact that these dimensions vary with q , indicate the multifractality of an experimental measure. The Berger-Parker index is introduced as a measure of dominance (Magurran 1988). For large q the right hand side of Eq. 4 is dominated by the largest values. Therefore, the dimension $D_\infty (= \lim_{q \rightarrow \infty} D_q)$ takes account of the scaling

property of this index of dominance (Caniego 2007).

Following Pielou (1975) the ratio D_1/D_0 may be interpreted as a measure of evenness in the context of multifractals (Caniego et al. 2003, 2006). The multifractal parameter $f(\alpha(0)) = D_0$ corresponds to the box-counting dimension and alludes to the scaling of the number of cells containing some pedotaxa abundance under successive finer partitions and then provides information about the size of the supporting set of pedotaxa abundances (not necessarily the whole interval of abundances). The entropy dimension $f(\alpha(1)) = \alpha(1) = D_1$ gauges the concentration degree of the distribution of abundances on the set supporting pedotaxa abundances whose geometrical size is characterized by D_0 . In this way D_0 could be interpreted as an indicator of richness of the pedotaxa distribution being the value of D_0 the maximum value D_1 may attain. The closer to one the ratio D_1/D_0 is, the more evenly pedotaxa are distributed over the set of sizes or abundances. This suggests that D_1/D_0 may be seen as a measure of the evenness of the distribution.

2. Pedodiversity and Biodiversity through Fractals and Multifractals

2.1 Simple scaling laws

In ecology the relation between the number of species S and the area A (e.g., of an island or analogous habitat of patchy spatial occurrence), may be described by two kinds of models: those that can be approximated by a linear relation between $\log S$ and $\log A$, i.e., $\log S = p \log A + q$, where p and q are constants (Arrhenius 1921, 1923, Preston 1962a, 1962b, MacArthur and Wilson 1967, May 1975), and those which approximate to a linear relation between S and $\log A$, i.e., $S = a \log A + b$ where a and b are constants (Gleason 1922, Fisher et al. 1943, Williams 1947, 1964). We emphasize that the first relation is a power function with exponent p and pre-factor e^q ($S = e^q A^p$) and the second is a logarithmic function of S in terms of A or alternatively an exponential function of A in terms of S . The former will be called the power model and the latter the logarithmic model. Each model of richness-area curve is related to a model of abundance distribution (i.e., the relation between abundance and the number of species possessing that abundance) (May 1975, Ibáñez et al. 2005). In particular the power law model for richness-area relationships is related to the lognormal distribution for the abundance distribution of the number of species. The view that the canonical lognormal model and the power function for species-area relationships are widely applicable, has become almost universally accepted (Connor and McCoy 1979, Sugihara 1981). However, for small samples, data usually fit simultaneously to both models of richness-area curves and they do not match lognormal models (Taylor 1978, May 1975, Coleman et al. 1982, Tokeshi 1993).

Much interest has been focused on the value of p , the slope of the log S versus log A regression line when dealing with the power law model under the assumption of the canonical lognormal distribution as a model for the distribution of abundances. For islands (mainland data, in general, give smaller p values than data from islands; e.g., see Rosenzweig 1995, 1998, 1999), linear regression leads to a slight overestimation of parameter p (e.g., $p = 0.262$ in Preston (1962a and b), and $p = 0.263$ in McArthur and Wilson (1967)) as compared to the asymptotically exact value of 0.25 (May 1975). Observed values often fall in the range 0.2–0.4 (Preston 1962a and b, McArthur and Wilson 1963, May 1975, Connor and McCoy 1979, Sugihara 1981, McGuinness 1984).

As explained above, following Harte et al. (1999), the power law of the richness-area relationship suggests scale invariance of the distribution of species which should be modeled with the self-similarity hypothesis. This pattern was observed for pedodiversity-area relationships in the Aegean archipelago (Ibáñez et al. 2005), in Europe (Ibáñez et al. 2009) and the whole World (Caniego et al. 2007), as described by the CORINE database of the European Union (Briggs and Martin 1988), the European Soil Database V2.0 (2004) and by FAO (1994), respectively. In all cases the exponent of the power law was close to 0.25 (Ibáñez et al. 2005, 2009, Caniego 2007).

Other simple scaling related to self-similar geometry has been observed. It is the Zipf law of rank-abundance list of pedotaxa for each continental landmass and the whole World (Caniego et al. 2007, Ibáñez et al. 2009). The rank-abundance list of pedotaxa is the list ordered by areal extent of soil types; the soil with the largest extension is the first in list. They followed the Zipf law: $e(r) \approx r^{-\alpha}$ where $e(r)$ is the areal extent of the soil type that occupies the rank r in the classification of pedotaxa by sizes. The value of α was about 1.7 and was greatest, 2.4, for Oceania, and lowest, 1.2, for the planet (Caniego et al. 2007). Thus, the Zipf law is less steep for Oceania. This suggests that soil distribution in Oceania is more uniform as compared to that for the continent as a whole. This is compatible with the behavior of other indicators of pedodiversity that will be explored below. It is worth noting that this rank-ordering technique conveys the same information as the distribution of pedotaxa sizes. In fact, the number of soil types with areal extent greater or equal than a certain value $N(e \geq \lambda)$ follows a Pareto distribution with exponent z , such that $z = 1 + (1/\alpha)$, i.e., $N(e \geq \lambda) \approx \lambda^{-z}$ (Caniego et al. 2007, Ibáñez et al. 2009). Similar patterns have been observed in the context of ecology (Hastings and Sugihara 1993).

To investigate the scaling behavior of the spatial distribution patterns of pedotaxa, the geographical extent of each particular soil type is mathematically idealized as a two-dimensional flat surface. The scaling behavior is statistically well-defined and allows estimation of the fractal dimension of pedotaxa in Europe (Ibáñez et al. 2009). These findings strongly

suggest the fractal nature of pedotaxa spatial distribution across Europe. Therefore, the pedotaxa fractal dimension may well play an important role when analyzing pedodiversity of earth soil systems.

2.2 A new approach to diversity distribution

Another important device when dealing with biodiversity analysis is through the investigation of the distribution of species abundances. May (1975) gives an analytic review of the subject in the context of ecology. Ibáñez et al. (2005) follows this methodology to investigate those relations in the context of pedodiversity analysis for islands. Standard statistical methods were used to gauge the abundance distribution models. The Smirnov-Kolmogorov test of goodness of fit indicated that the log-series and the broken stick distribution models should be rejected, whereas the geometric and log-normal models were not rejected ($P < 5\%$). The fact that the geometric model cannot be excluded as a model for the distribution of pedotaxa sizes may suggest that non-purely random factors may become significant. The log-normal model is associated with pure random processes or, alternatively, as soon as several factors play a major role this model reflects the statistical Central Limit Theorem (May 1975).

A different approach to deal with distributions of abundances in both biodiversity and pedodiversity analysis comes from the interpretation of power scaling laws observed in nature as a by product of self-similarity, as we argued above. This self-similarity expresses the deterministic chaotic behavior of natural environmental systems. From this view, Caniego et al. (2006) explored the multifractal features of pedotaxa distributions at the planetary scale. Singularity exponents $\alpha(q)$, fractal dimensions $f(\alpha(q))$ and Rényi dimensions D_q with the corresponding coefficients of determination R^2 for each pedotaxa-abundance distribution were estimated for the continents and the World. The singularity behavior of the pedotaxa-abundance distributions was clearly established by the noticeable variation of all the multifractal parameters with q . Multifractal and Rényi spectra had standard shapes, similar to multinomial measures. Therefore, they could be simulated with those kind of multifractals having as support fractals sets (Cantor-like sets could be the simplest choice) in order to take account of the fact that the box-counting dimension D_0 (Eq. 4 for $q = 0$) is always less than 0.949. In the case of Europe, the spectra shapes are almost symmetric. In the other cases, they are asymmetric with the right branch longer than the left one. The World represents the extreme of this behavior. Africa and Oceania correspond to a situation in the middle of these two extremes. Then, the features of the singularity spectra of multinomial measures suggest that the self-similarity of the abundance distribution of Europe ought to have a pattern close to binomial multifractals, while the others might have a structure closer to

trinomial or more general multinomial measures. This analysis indicates that the complex behavior of pedotaxa-abundance distributions at the planetary level has a well-defined multifractal structure. Being multifractal patterns, natural ingredients in chaotic phenomena of nonlinear dynamics, these results may be interpreted as a positive test reinforcing the suitability of the theory of multifractals to analyze pedodiversity patterns of Earth surface systems.

The maximum number of dyadic partitions used for parameter estimates is related to the quality of the fit. This number establishes an effective lower cut-off in the range of possible considered scales. Actually, this lower cut-off is related to the nature and precision of the dataset. But the multifractal analysis determines also an upper cut-off in the range of scales where a good fit is detected. In this way, a range of scales may be selected where a scaling is depicted and a multifractal behavior is observed. Let us consider Asia as a case study. Caniego (2007) observed that the spectrum of Rényi dimensions did not follow the pattern of multifractal measures, when all the possible soil types in this continent were considered (Fig. 4.7A). The shape of the D_q function changed dramatically when the soil with the greatest areal extent was not included (Fig. 4.7B) and it kept changing when more soil types were excluded (Fig. 4.7C). This trend stabilized when the number of considered soils was 110 (there were 128 different soil types in Asia) (Fig. 4.7D) and it coincided with the best coefficient of determination of

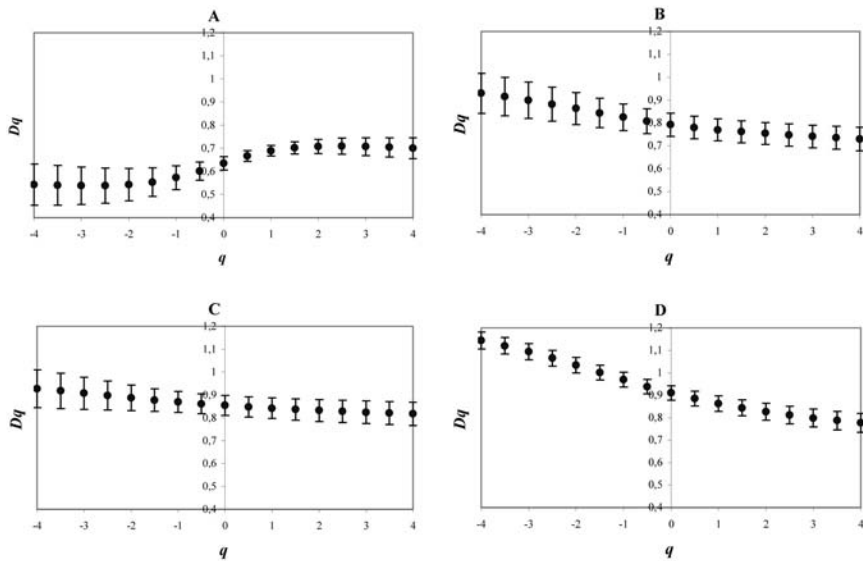


Figure 4.7 D_q functions for Asia with different number of soils with indication of extension in km^2 of the largest soil considered: A) 128 soils 36,623 km^2 , B) 127 soils 12,766 km^2 , C) 126 soils 10,582 km^2 , D) 110 soils 3,868 km^2 .

D_0 . This was the criterion used to determine an upper cut-off in the pedotaxa distribution of sizes to perform dimension estimates for the multifractal analysis of pedodiversity. In this way a characteristic scale may be selected in each case. About half of the taxa whose sizes were larger than the upper cut-off selected with this criterion corresponded to soil units of the major groups Regosols and Lithosols, or to miscellaneous units such as Dunes and Shifting Sand, Rock Debris and Glaciers (FAO 1974).

Similar effects have been reported when considering fractal patterns at a very large scale, as is the case of the characterization of the distribution of galaxies in astrophysics (Gaite et al. 1999). In this context, the upper cut-off is understood as the transition to homogeneity. In a similar way this break of the multifractal behavior of the pedotaxa abundance distribution might be interpreted as a transition to a less complex pattern. In this respect it is interesting to consider the reflections of Mandelbrot (1977) on the effective dimension of an object. The mathematical theory provides a model where scaling behavior is valid for any scale, i.e., any real number, no matter how large or small it might be. When analyzing real objects we need to consider the resolution and the actual size of the representation of the object. They provide the minimum "available" size but also the maximum "possible" one and can be seen as the boundaries of scales where the mathematical model captures the multifractal pattern of the real object. Near these boundaries the scaling could be unpleasant as smaller (or greater) scales cannot be explored, but the mathematical model needs them in order to produce its outputs. These limits may be related to the incompleteness of the dataset as this does not contain details that are effectively available. This might suggest an explanation of the lower cut-off in this case. One can imagine a finer dataset with more detailed information for smaller scales. But, as pointed out by Mandelbrot (1977), they may also be related to a transition between scales with well-defined dimensions. In this way the upper cut-off might be understood as the appearance of different pedological patterns for larger scales.

2.3 Indices of diversity in fractal environments

The behavior of richness R , entropy H , Simpson index S , Berger-Parker index B and evenness E , and their multifractal counterparts, D_0 , D_1 , D_2 , and D_∞ , and the ratio D_1/D_0 , were explored at planetary level from the most detailed available global dataset based on the second level of the FAO 1974 classification units (Caniego et al. 2007). Figure 4.8 depicts the correlations of diversity indices and Rényi dimensions. Logarithmic axes have been chosen for richness R , Simpson index S and Berger-Parker index B to take into account the absence of logarithms in the definition of these indices. The value of D_∞ was approximated by D_4 . It was the

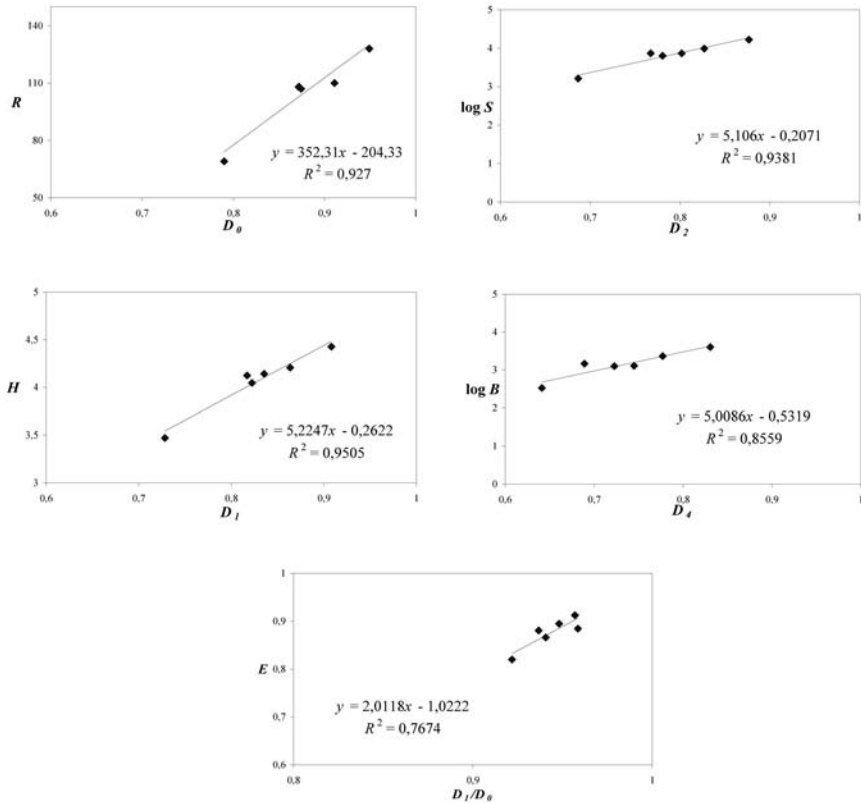


Figure 4.8 Correlations between diversity indices and Rényi dimensions. Richness (R), Entropy (H), Simpson (S), Berger-Parker (B) and Evenness (E).

greatest q that was estimated. These correlations were anticipated by the theory and could be interpreted, when a suitable scaling behavior is identified, as an indirect indication of the suitability of this mathematical tool for describing pedodiversity in fractal environments. In the context of complex and multifractal phenomena, these findings suggest that Rényi dimensions D_0 , D_1 , D_2 , and D_∞ , and the ratio D_1/D_0 would be interpreted as indicators of richness, heterogeneity, correlation, dominance and evenness, respectively.

It is worth noting that D_0 , D_1 and D_1/D_0 increase as the area of the continents increases. This feature suggests that area is the major driving force. The ratio D_1/D_0 is rather uniform for all continental landmasses suggesting that the evenness for all of them is very similar and close to the value obtained at the planetary level. The box-counting dimension D_0 follows the same trend but the role of the area is reinforced for the ratio

D_1/D_0 . The entropy dimension D_1 shows a more pronounced behavior, indicating that diversity is more affected than richness and evenness by area. It is worth noting here that in the pedodiversity literature other hypotheses have been put forward to explain habitat heterogeneity. It is hypothesized that habitat heterogeneity and pedodiversity are largely driven by plate tectonic dynamics and their repercussions in Earth surface systems (Ibáñez and Effland 2011). Even if similar regularities have been observed for biodiversity and pedodiversity patterns this hypothesis suggests that different underlying assumptions are needed to explain these similar patterns (Rosenzweig 1995).

The box-counting dimension D_0 ranges from 0.79 (Oceania) to 0.949 (The World). This reflects the fact that the abundances of pedotaxa in the whole World are located in a larger amount of cells than in any continent for the considered partitions. The entropy dimension D_1 ranges from 0.728 (Oceania) to 0.908 (The World). These results suggest that both richness and diversity confer similar order to these distributions. Richness as measured by D_0 establishes the following order: Oceania, Europe, America, Africa, Asia, and The World, while diversity as measured by the entropy dimension D_1 yields: Oceania, Europe, Africa, America, Asia, and The World. It is worth noting that the changes correspond to continents with very similar richness and diversity. So, it is quite accurate that with increasing richness, diversity increases too. High values of evenness as measured by D_1/D_0 were obtained. They were greater than 0.922 (Oceania). The same regularities as before were found. In this case, the order that D_1/D_0 establishes was Oceania, Europe, Africa, Asia, America and The World. Only America and Asia are changed with respect to the order established by D_1 . Thus, it might be said that the increase of any of these indices brings about an increase of the others for this scenario. However, further investigations are needed in order to validate this hypothesis.

3. News for the Near Future

Fractals and multifractals are here to stay. They provide well-behaved tools for pedodiversity and biodiversity analyses. The suitability of these new techniques stems from the very nature of this subject and the essence of the matter to which it belongs: geo- and bio-science. But, they also offer a new theoretical framework to deal with complexity in Nature. These days, it is not possible to look through the mathematical and physical theories of nonlinear dynamics and chaos in order to investigate the complex nature of geometric irregularities or apparently random phenomena.

The fractal dimension supplies an established tool to parameterize geometrical complexity. But, new indicators of geometrical attributes are needed to take account of the differences observed between geometrical

objects with the same fractal dimension. This reflects the fact that, length and connectedness provide a way to parameterize a one dimensional regular set of points, while area, length of the boundary and connectedness do the same work in two dimensions. In fact, the Minkowski functionals of the theory of Integral Geometry furnish an interesting characterization of regular geometrical forms in any integer dimension (Michielsen and De Raedt 2000). In this respect some new fractal parameters have been put forward to fill this gap. Two examples are lacunarity (Mandelbrot 1977, Plotnick et al. 1996) and local geometry entropies (Andraud et al. 1997). But, we are far from having a complete description of the geometrical features of fractal shapes similar to the above mentioned for integer dimensions.

New tools in the realm of multifractals to address new problems or to give alternative ways for estimating multifractal parameters have been introduced. This is the case of the multifractal detrended fluctuation analysis (Kantelhardt et al. 2002) to investigate non-stationary processes, the joint multifractal theory (Meneveau et al. 1990, Kravchenko et al. 2000) for the simultaneous analysis of several multifractal measures with the same geometric support and quantify the relationships between them or, the wavelets representation of multifractals (Muzy et al. 1991) that provides a multiscale decomposition of measures. Turiel et al. (2006) provide a broader perspective on this question. This listing does not pretend to exhaust the possibilities that have been explored up to this date in the literature, but rather to give a flavor of the new possibilities that multifractal analysis can offer to pedodiversity and biodiversity analyses.

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CHAPTER 5

Pedodiversity and Landscape Ecology

Asunción Saldaña

1. Introduction

Soil is an essential and indispensable component of ecosystems (Yaalon 2000). Hence, it is obvious that biodiversity conservation is impossible without simultaneous conserving of the environment of biological organisms and taking into account soil diversity, structure and soil cover pattern in particular landscapes (Amundson et al. 2003, Decaëns 2010, Ibáñez et al. 2008, Khaziev 2011). Given that the spatial component of the environment is crucial, new approaches to soil patterns are necessary for appropriate landscape planning, management and conservation. The integration of landscape ecology fundamentals together with soil science principles can be helpful in this regard.

Ecology comprises the study of the interrelationships between organisms and their surroundings (Ricklefs 1973). Ecological organization ranges from individuals to the biosphere and includes populations, communities, ecosystems, landscapes and regions. Accordingly, autecology, population ecology, community ecology, systems ecology, landscape ecology, regional ecology and global ecology are all branches of ecology. Therefore, landscape ecology focuses on an organizational level above that addressed by community and ecosystem ecology.

This chapter deals with the relationship between pedodiversity and the theory and methods of landscape ecology. The result of the integration of both approaches is applied to soilscape-pattern analysis. The chapter is structured in three parts: the first one focuses on landscape ecology concepts and how they can help in the analysis of the environment; the second part concentrates on the application of previous ideas to the analysis of the complexity of the soilscares, i.e., the pedologic portion of the landscape (Buol et al. 1973, Hole 1978); and the third part concerns the conservation of pedodiversity.

2. Landscape Ecology

Although the definition of landscape ecology has been dealt with extensively in landscape ecological literature, there remains confusion among other scientists as to exactly what landscape ecology is and, particularly, its unique contribution to ecology as a whole (Fahring 2005).

2.1 Origins

As with many of the natural sciences, the study of landscapes has its origins in the 19th-century scientific explorations by Europeans (e.g., A. Von Humboldt). Although the primary aim was to describe and categorize different environments and vegetation types, some attention was also given to the environmental determinants and consequences of human occupancy of these places. Some of these natural scientists also began to seek underlying principles to explain the broad-scale spatial variations in environments and the distributions of plants and animals (Wiens et al. 2007). This is what Forman (1995) refers to as the initial phase of landscape ecology, “the natural history and physical environment phase”. Two Russian scientists of this period, L.S. Berg and N. Solnetsev, defined geographic landscape as “a genetically homogeneous territory in which a regular recurrence of the same interrelated combination of factors takes place; these factors are geological composition, forms of relief, surface and subterranean waters, microclimates, soil units, phytocenoses, and zoocenoses”. A. Watt added a spatial component to the study of plant communities. In North America, C. Sauer highlighted the relationship between natural and cultural elements (references in Wiens et al. 2007).

The second phase, from about 1950 to 1980, set the stage for today’s landscape ecology. The term landscape ecology was introduced by the German bio-geographer Carl Troll. His 1950 paper is generally regarded as the foundation paper for landscape ecology. In this publication the term *landschaftsökologie* (landscape ecology) was first extensively used, although Troll had already coined the term earlier. In it he joined the ideas from

Russian geographers with Tansley's ecosystem concept (proposed in 1935). His background in natural sciences and his work in mountainous areas gave him the awareness of the interdependence among soils, slope and vegetation. Besides, experience in the use of aerial photography allowed him to focus on terrain analysis at a scale that was relevant to regional planning. Another contributor to landscape ecology in this period was C.S. Christian whose land system concept was useful for comprehensive surveys and land classification of broad regions. Tricart and Kilian used an eco-dynamics approach to landscape study based on its dynamic behaviour (Tricart and Kilian 1982). R. Whittaker and J. Curtis showed that environmental and disturbance gradients of varying steepness can create community patchiness in space with varying species composition and contrast (Wiens et al. 2007 and references therein).

The third phase, after the 1980s, has resulted in the emergence of the overall conceptual design of landscape and regional ecology (Forman 1995). Scientists focus on landscape dynamics, description and quantification of landscape pattern and planning and management (including conservation). Some early scientists of this period are R.T.T. Forman, M. Godron, J.A. Wiens and M.G. Turner among others.

Landscape ecology can then be defined as a subdiscipline of ecology that emphasizes the interaction between spatial pattern and ecological process, that is, the causes and consequences of spatial heterogeneity across a range of scales (Turner et al. 2001). Specifically, it considers four aspects of landscape systems: (1) the evolution and dynamics of spatial heterogeneity—how the landscape mosaic is created and how it changes; (2) the interactions between, and exchanges of materials across, heterogeneous landscapes—how materials and organisms move from one patch to another; (3) the influence that spatial heterogeneity of the landscape mosaic has on biotic and abiotic processes in the landscape; and (4) the management of spatial heterogeneity (Turner 1989).

Due to its history, landscape ecology crosses several disciplines (geography, ecology, landscape planning, etc.). So it can be considered as a *transdisciplinary* science (Naveh and Lieberman 1984, Zonneveld 1990) or an interdisciplinary area of research (Risser et al. 1984) because it is not just a combination of the methods of various sciences but integrates landscape components on a higher level. More recently, Wu and Hobbs (2002) indicated that landscape ecology is a multidisciplinary science, and that collaboration within and beyond the realm of ecology is needed to make it a truly interdisciplinary or transdisciplinary. The diversities in approach and traditions in landscape analysis (Turner 2005) are both contrasting and complimentary (Wu and Hobbs 2002) and an inherent part of the field. One approach more anthropocentric, has been used in Europe and relates to landscape planning. The second approach encompasses the causes and

consequences of spatial patterns at different scales that are defined by the organism or process of interest and reflects traditions in North America and Australia. It is remarkable that during this time, more attention was devoted to the biotic part of the landscape than the abiotic component. However, soils received attention in the European eco-geographical school (e.g., Troll, Tricart, etc.). Soils have been considered in explaining the occurrence or behaviour of organisms, i.e., soils were considered as a background of the ecosystem but not as an inherent part of the system (Klink et al. 2002).

2.2 Landscape and its elements from an ecological point of view

Whereas portions of a region from an ecological point of view are quite dissimilar, a landscape manifests an ecological unity throughout its area. Within a landscape several attributes tend to be similar and repeated across the whole area, including landforms, soil types, vegetation types, local fauna, natural disturbance regimes, land uses, and human-aggregation patterns. Thus, a landscape is a heterogeneous land area composed of a cluster of interacting ecosystems that is repeated in similar form throughout (Forman and Godron 1986).

According to Forman (1995) landscape elements are spatial elements or homogeneous units at the landscape scale whereas the spatial elements within regions are landscapes. They are made up of individual trees, shrubs, herbs, and small buildings. Landscape ecologists recognize three basic spatial elements, including patches, corridors, and matrix; between these elements there are more or less distinct boundaries or ecotones. These elements form a mosaic at any scale (Forman and Godron 1986):

- (1) *Patches*: these are relatively homogeneous non-linear areas that differ from their surroundings. They are the basic pieces of landscape mosaics. Urban settlements or a fragmented forest within an agricultural matrix area are some examples. In the field of pedology, Alfisols and Ultisols embedded in a matrix of Entisols are located on the southern slopes of the Ayllón range, central Spain (Ibáñez et al. 1987). In the same way, patches of consociations of Typic and Calcic Palexeralfs are surrounded by consociations of Typic and Calcic Haploxeralfs in the middle-high terraces of the Henares River, central Spain (Saldaña 1997). Patches have familiar attributes such as size, shape and the nature of the edge. These attributes in turn have widespread ecological implications for productivity, biodiversity, soil and water. Some internal microheterogeneity is possible but it must be repeated in similar form throughout the area of a patch.
- (2) *Corridors*: these are strips of a particular type that differ from the adjacent land on both sides. Streams, hedgerows, roads, pipelines, a

mountain range are examples of corridors. One example in the field of pedology is the association of Typic Xerofluvents and Fluventic Haploxerepts found in the swales and vales resulting from the dissection of the Henares River terraces, central Spain (Saldaña 1997). Width, connectivity, curvilinearity, narrow, breaks, and nodes control their important functions, including conduit, barrier and habitat.

- (3) *Matrix*: this is the background ecosystem or land-use type in a mosaic, characterized by extensive cover, high connectivity, and/or major control over dynamics. For example, in a large contiguous area of mature forest, embedded with numerous small disturbance patches (e.g., timber harvest patches), the mature forest constitutes the matrix element type, because it is greatest in areal extent, is mostly connected, and it exerts a dominant influence on the area flora and fauna and ecological processes.

Landscape elements are generally identifiable in aerial photographs and often range from around 10 m to 1 km or more in width (Forman and Godron 1986). Actually, the size of a landscape is not fixed *a priori*, because it depends on the process or the organisms under focus (Farina 2000). Processes that operate in a landscape across a broad range of spatio-temporal scales and which, in turn, influence many landscape patterns include disturbance (natural and human), fragmentation, connectivity and water and nutrient flows across landscapes.

2.3 What makes landscape ecology different from other branches of ecology?

Two important aspects distinguish landscape ecology from other branches of ecology:

- (1) *Spatial component and its mutual relationship with ecological processes*. The main focus is on three characteristics of the landscape (Forman and Godron 1986, Risser et al. 1984):
 - Structure refers to the spatial relationships between distinctive ecosystems (i.e., the distribution of energy, materials, and species in relation to the sizes, shapes, numbers, kinds and configurations of components). This aspect has been largely ignored in ecology mainly because of the perceived difficulty of conducting broad-scale studies (Fahring 2005).
 - Function refers to the interactions between the spatial elements, that is, the flow of energy, materials and organisms among the component ecosystems. Such considerations had already been included in Tricart and Kilian's morpho-pedogenetic approach to landscapes (Tricart and Kilian 1982).

- Change refers to alteration in the structure and function of the ecological mosaic through time.
- (2) *Scale*. Because landscape ecologists deal with processes occurring at a wide range of spatial and temporal scales, it is important to carefully define both spatial and temporal scale because it influences the conclusions drawn by an observer and whether the results can be extrapolated to other times and locations (Turner et al. 2001). There is no “best” scale at which to study the environment; the appropriate scale depends on the research question at hand (Noss 1992) so that an exploration of a broad spectrum of spatial and temporal scales is recommended (Wiens et al. 1986). Scale has a prominent role in landscape ecology because (see Turner et al. 2001):
- Recent issues related to the environment were manifested over large areas (acid rain, global climate change, habitat fragmentation, conservation of biodiversity, etc.). Therefore, extrapolation from fine to broad scales has become crucial for ecosystem management.
 - Recognition that biological interactions in the environment occur at multiple scales. In this sense, ecologists became aware that the spatial and temporal scales important to humans were not necessarily the scales that were relevant to other organisms or a wide range of ecological processes.
 - Hierarchical structure in nature and positive correlation in spatial and temporal scales of a varying process. In addition, a new mathematical theory such as fractals (Mandelbrot 1983) seemed to explain some complicated patterns in nature, while offering potential for the development of rules that would allow observations to be transferred from one scale to another.

The notion that ecosystems are not isolated systems and cannot be understood without considering the flow of energy and material across ecosystem boundaries is crucial. New ideas about the heterogeneity and the role of disturbance regime (Pickett and White 1985) in ecological process represent further progress, including the hierarchy theory, connectivity concepts, metapopulation theory, the percolation theory and the information theory (Farina 2006). Regarding the analysis of spatial scale, there is a growing variety of tools (Turner et al. 2001 and references therein), including correlograms, semivariance analysis, lacunarity analysis, spectral analysis and a variety of fractal-based methods.

Wu and Hobbs (2002) drew attention to several points as the main lines of research of landscape ecology: (1) our understanding of the reciprocal relationship between spatial pattern and ecological flows or processes is incomplete. While much of the attention has been given to spatial pattern analysis, research emphasis should be directed towards processes

themselves and how they affect, and are affected by, landscape pattern, using an integrative approach (i.e., including not only fauna and vegetation, but also landforms, soils, etc.). (2) More research is needed to understand the causes, processes and ecological consequences of land use and land cover change. (3) To effectively deal with the complexity of landscapes, insights from the science of complexity and non-linear dynamics may play an important role. (4) While scale effects are widely recognized in landscape ecology, questions are yet to be addressed on how to determine appropriate scales for understanding particular patterns and processes and how to scale up or down across heterogeneous landscapes. (5) The lack of replication in landscape studies causes problems when using traditional scientific methods that hinge primarily on experimentation. (6) A sound technical and ecological understanding of the large number of landscape metrics already developed is still lacking. (7) The emphasis on “natural” landscapes is slowly but steadily giving way to a perception of the importance of humans in shaping the landscape. (8) It is important to address the questions of landscape-pattern optimization (e.g., optimization of land use pattern, optimal landscape management, optimal landscape design and planning). (9) The application of landscape ecological principles in biodiversity conservation and maintaining the sustainability of landscapes. (10) Despite the availability of technological advances in surveying and information-processing, there are problems concerning the acquisition, quality and analysis of landscape data.

2.4 Quantification of landscape pattern

Ecological systems are spatially heterogeneous, exhibiting considerable complexity and variability in time and space (Gustafson 1998). This pattern is visible especially at the landscape scale. This section describes geometrical attributes (i.e., metrics) of landscapes, although the study of the landscape also requires additional tools such as spatial statistics, Geographic Information Systems, Remote Sensing techniques and Global Positioning Systems (Farina 2006).

The mechanisms that create heterogeneity (mosaic pattern) include substrate heterogeneity such as different soil types causing vegetation patchiness; natural disturbance; and human activities such as ploughing and building (Forman 1995). Therefore, in a rapidly changing world, monitoring and analyzing landscape dynamics is more necessary than ever.

One of the trademarks of landscape ecology has been its extensive use of landscape metrics for spatial pattern analysis. A large array of features and measures can be derived from a map or image of a landscape. Some of these indices have proved useful for the description of landscape structure and spatial dynamics over a broad range of spatial and temporal scales. Some

of the measures available in literature, more than 100, such as patch size or shape, nearest-neighbour distance, or perimeter: area ratio, portray features of particular patch types independent of their surroundings. Others, such as adjacency or contrast, deal explicitly with what lies across the boundaries of a given patch type. Still other measures—semivariance, lacunarity, fractal dimension, patch diversity, connectedness, or various indices of heterogeneity, for example—characterize features of the mosaics as a whole (e.g., Arnot et al. 2004, Baldwin et al. 2004, Herold et al. 2005, Forman and Godron 1986, Krummel et al. 1987, O'Neill et al. 1988). The importance of scale-dependent variations has also been tested using the above indices to describe landscape characteristics (Baldwin et al. 2004, Buyantuyev and Wu 2007, Corry 2005, Turner 1990, Wu 2004). Table 5.1 shows a selection of indices of wide use in landscape ecology. Many of them have already been implemented in commercial or free of charge software.

Several authors (e.g., Wu and Hobbs 2002, Haines-Young 2005) warn that most contemporary work on pattern has focused on the analysis or description of spatial geometry and has failed to explore relationships between pattern and process. Landscape indices and map data are sometimes used without testing their ecological relevance, which may not only confound interpretation of results, but also lead to meaningless results. This tendency has been exacerbated by the availability of digital landscape data and GIS algorithms that allow the calculation of a whole range of landscape metrics. However, Li et al. (2005) showed that none of the available indices is appropriate for all aspects of a landscape pattern whereas Wu (2004) highlighted the need for multiscale analysis in order to adequately characterize and monitor landscape heterogeneity. Some of these indices (e.g., diversity index, fractal dimension) may also be used to explain the relationships between soil pattern, soil forming processes and landscape evolution.

Examples of landscape and soilscape metrics applied to soil-geoform units in the Jarama-Henares interfluvium and the Henares River valley (central Spain) (Fig. 5.1) can be found in Saldaña (1997) and Saldaña et al. (2011). The objective of these studies was to develop a landscape evolution model of the area during the Plio-Quaternary. To do so, a qualitative description of the patterns was carried out following the steps proposed by Fridland (1976) and Hole and Campbell (1985). Next, quantification of the geo-pedologic combinations present in the maps at regional (1:50,000) and local (1:18,000) scales was done using a set of 22 indices. Features widely used in landscape ecology (i.e., fragmentation, dominance, geo-pedologic unit diversity, relative spatial diversity, size and shape, neighbourhood and interaction) were analyzed. Similarly, Saldaña and Ibáñez (2004) assessed the taxonomic pedorichness and pedodiversity on plot maps at 1:100 scale.

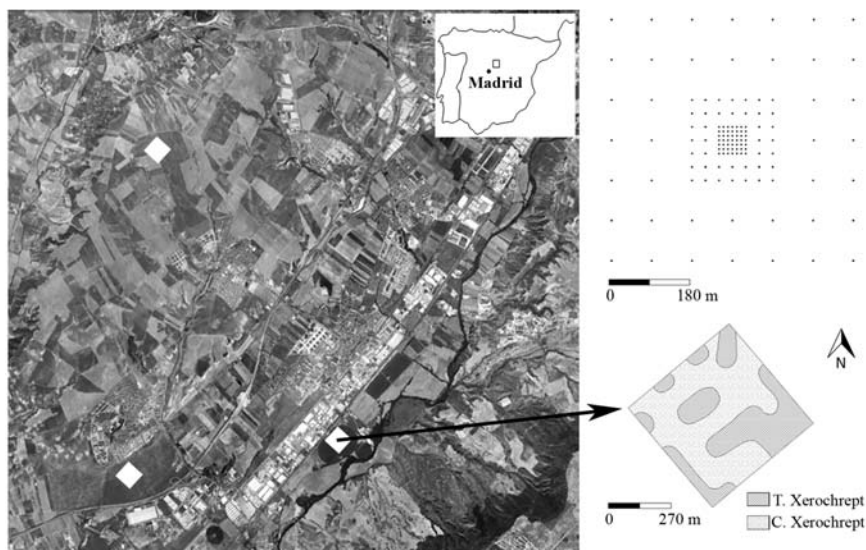


Figure 5.1 Sample areas, sampling scheme and one soil map at plot scale in the Jarama-Henares interfluvium and the Henares River valley (modified after Saldaña et al. 1998).

According to these authors, diversity indices can be used as indicators of soilscape evolution, while shape indices, in particular the fractal dimension, are useful indicators of terrain stability and relief dissection.

Another important finding regarded the effect of scale in the analysis: the values of the indices were higher and the number of indices required to describe appropriately the soilscape patterns were smaller at the local than at the regional scale. Another conclusion was that not all indices were fully efficient and a subset should be chosen from the indices commonly applied so that they can be useful for characterizing soilscales at different scales.

3. Structure of the Soil Mantle in the Landscape

The soil continuum presents a pattern characterized by a more or less regular distribution of soil bodies in space that are associated with bodies of non-soil (Buol et al. 1973). The soil cover pattern is distinguished from the zonal or regional soil pattern that is expressed by a gradual change in soil over large areas, resulting from climatic gradients (Fridland 1976). The differences among soils generate local heterogeneity, but the repetition of these patches of local heterogeneity gives the soil a regularity of structure.

The idea of a structured soil mantle was developed by several Russian pedologists such as Sibirtsev, Dokuchaev, Ivanova, Gerasimov and Fridland (Fridland 1976). The latter made a significant contribution to this approach

Table 5.1 Metrics for landscape pattern analysis.

Type of metrics	Index	Formula	Variables	Source
Diversity indices	Richness	$S = \sum n_i$	S = Richness n_i = Number of different community types present	Romme (1982)*
	Evenness	$E = \frac{H'}{\ln(S)}$	E = Evenness H' = Proportion of total landscape covered by i^{th} community type S = Richness	Pielou (1975),** Romme (1982)*
	Diversity	$H' = -\sum_{i=1}^m p_i \ln(p_i)$	H' = Measure of diversity p_i = Fraction of the landscape occupied by land use i m = Number of land use types in the area	Shannon and Weaver (1949),* Baker and Cai (1992)***
	Dominance	$D = \ln(m) + \sum_{i=1}^m p_i \ln(p_i)$	D = Dominance index p_i = Fraction of the landscape occupied by land use i m = Number of land use types in the area	Pielou (1975),** O'Neill et al. (1988)**
	Fragmentation	$FI = \frac{(n-1)}{(c-1)}$	In a GIS environment: FI = Fragmentation index N = Number different classes C = Number of cells considered in a 3 X 3 window (i.e., $c=9$)	Eastman (1993)***
Neighbourhood and interaction indices	Relative patchiness	$RP = \frac{\sum_{i=1}^N D_i}{N} * 100$	RP = Relative patchiness index D_i = Dissimilarity value for the i^{th} boundary between adjacent cells N = Number of boundaries between adjacent cells in a watershed	Patton (1975),* Romme (1982)*
	Isolation of a patch	$r_i = \frac{1}{n} \sum_{j=1}^{j=n} d_{ij}$	r_i = Degree of isolation of the unit i n = Number of neighbouring map units d_{ij} = Distance between the unit i and any adjacent unit j)	King (1969),* Forman and Godron (1986)

Neighbourhood and interaction indices	Accessibility of a patch	$a_i = \sum_{j=1}^{j=n} d_{ij}$	a_i = Index of accessibility of patch i d_{ij} = Distance along a linkage between patch i and any of the n neighbouring patches j	Lowe and Moryadas (1975),* Forman and Godron (1986)
	Interaction among patches	$l_i = \sum_{j=1}^{j=n} \frac{A_j}{d_j^2}$	l_i = Degree of interaction of the patch i with n neighbouring patches A_j = Area of any adjacent patch j d_j = Distance between the edges of the patch i and any patch j	Whitecomb et al. (1981),* Forman and Godron (1986)
	Dispersion of patches	$R_c = 2d_c \left(\frac{\lambda}{\pi} \right)$	R_c = Index of dispersion d_c = Average distance from a patch to the nearest neighbouring patch λ = Average density of patches	Pielou (1977),* Forman and Godron (1986)
	Contagion	$C = 2n \ln(n) + \sum_{i=1}^n \sum_{j=1}^n p_{ij} \ln(p_{ij})$	S = Number of habitat types p_{ij} = Probability of habitat i being adjacent to habitat j	Godron (1966),* O'Neill et al. (1988)**
	Juxtaposition	$J = \sum_{n=1}^s \frac{q_n w_{ij}}{12}$	J = Juxtaposition index q_n = 2 if cell n is horizontally or vertically adjacent and q_n = 1 if cell n is diagonal w_{ij} = Weight to be given to edges between patches m_i and m_j	Baker and Cai (1992)***
	Centre Versus Neighbours	-	-	Murphy (1985) (reference in Eastman 1993)***
	Edges	$E_{ij} = \sum e_{ij} * l$	E_{ij} = Edges index e_{ij} = Number of horizontal and vertical interfaces between cells of types i and j l = Length of the edge of a cell	Turner (1987)**

Table 5.1 contd....

Table 5.1 contd....

Type of metrics	Index	Formula	Variables	Source
Size and shape indices	Patch size	-	-	Forman and Godron (1986)
	Patch size standard deviation.	-	-	Forman and Godron (1986)
	Shape of a patch	$D = \frac{P}{2\sqrt{\pi A}}$	D = Shape index P = Perimeter of a patch A = area	Patton (1975),* Forman and Godron (1986)
	Fractal dimension	$A = kP^{2/D}$	A = Patch area P = Patch perimeter D = Fractal dimension	Burrough (1986),** Krummel et al. (1987)**
Edge indices	Perimeter	-	-	McGarigal and Marks (1995)
	Total edge	-	-	McGarigal and Marks (1995)
	Edge density	-	-	McGarigal and Marks (1995)

*: Reference available in Forman and Godron (1986); **: Reference available in Turner (1989); ***: Reference available in Saldaña (1997)

especially after publishing “Pattern of the soil cover” (1972), originally in Russian and translated into English in 1976.

Fridland defined soil cover as “the entirety of soils occurring on a territory; this is a three-dimensional body with its horizontal and vertical extents, respectively equal to the area and depth of the soils on the territory”. He called soil formation free from any internal pedogeographical boundary and with variable size Elementary Soil Areal (ESA) (Fridland 1976). The ESA represented a body of soil that belongs to a certain classificational unit of the lowest rank and occupies space which is bound on all sides by other ESAs or non-soil formations. So an ESA would correspond to a polypedon. The ESAs could be homogeneous, consisting of similar pedons, or heterogeneous, formed by contrasting soil pedons. The ESAs would group to constitute soil combinations, which were formed by spatially and genetically related ESAs. Fridland defined six types of soil combinations: combine, complex, variation, patchiness, mosaic and tached (Table 5.2). An Elementary Soil Areal was characterized by:

- Its content which is controlled by the classification-position of its constituent soil or soils.
- Its geometry, which is controlled by its size, shape and contour irregularity. The maximal sizes are very large, running into thousands of hectares.
- Its place in soil combinations, regarding neighbourhood issues.
- Its ecological characteristics, which describes the conditions leading to its formation and helps understand the causes for its segregation and for the formation of its boundaries along every one of its neighbours.

The genetic and geometric forms of the structure of the soil mantle depend on the factors of soil formation, but lithological-geomorphological conditions are expressed most clearly in them. In this regard, man’s activity would be also included as a factor differentiating the soil mantle (Fridland 1974, Hole 1978). Fridland (1976) proposed or collected from other authors several indices to describe his soil combinations (Table 5.3). It is interesting that heterogeneity was already considered as a property to describe soil cover.

Hole and Campbell (1985) reviewed and updated Fridland’s work from an interdisciplinary point of view. Soilsapes are multi-polypedonic units delineated at different scales, and, at whatever scale of generalization, may be characterized by their internal make-up and by their relation to surrounding soilsapes. It is worth mentioning that these authors considered the landscape to have a pedologic structure (pattern, which means that it was characterized by their size, shape and arrangement of component soil bodies).

Table 5.2 Types of soil combination (Saldaña 1997).

Soil combination	Pedologic contrast	Relief variations	Origin	Size of the ESA	Genetic links among components	Regular recurrence of components?
Combine	High	Mesorelief*	Relief, parent rock, etc.	Large enough to be farmed separately	Strong	Yes
Complex	High	Microrelief**	Relief, parent rock, etc.	Too small to be farmed separately	Strong	Yes
Variation	Low	Mesorelief*	-	-	Strong	Yes
Patchiness	Low	Microrelief**	-	-	Strong	Yes
Mosaic	High	-	-	Small	Weak	No
Tachet	Low	-	Biological	Small	Weak	No

* 10–100 m; ** < 10m

Table 5.3 Some metrics for soil landscape analysis based on Hole and Campbell (1985).

Type of metrics	Index	Formula	Variables	Source
Density	Count of mean number of soil boundaries intersected by a transect of unit length	-	-	Hole and Campbell (1985)
	Count of nodes (i.e., junctions of soil boundaries) per unit area	-	-	Hole and Campbell (1985)
	Mean density of soil bodies	$M_d = N_d / A$	M_d = The mean density of soil bodies N_d = The number of soil types A = The total area (in km ²)	Hole and Campbell (1985)
	Length of drainage ways per unit area	Total length of stream drainageways per 1 square meter	-	Hole (1983)*
Composition (richness)	Count of soil bodies	-	-	Hole and Campbell (1985)
	List of pedotaxa and non-soil units (i.e., pedorichness)	-	-	Hole and Campbell (1985)
	Proportionate extents of components	-	-	Hole and Campbell (1985)
Diversity	Soil body size distribution	-	-	Hole and Campbell (1985)
	Number of soil map legend units per unit area	$M_{su} = N_{su} / A$	M_{su} = Map soil unit index N_{su} = Number of soil map legend units A = Area	Hole and Campbell (1985)
	Index of heterogeneity	$HI = M_d * n$	HI = Heterogeneity index M_d = Mean density of soil bodies n = Number of soil map units present in the area	Hole and Campbell (1985)
	Number of soil landscape positions	-	-	Hole and Campbell (1985)
	Soil moisture regime diversity	-	-	Hole and Campbell (1985)

Table 5.3 contd....

Table 5.3 contd....

Type of metrics	Index	Formula	Variables	Source
Size and shape	Soil body shape index	$SI = \frac{P}{2\sqrt{\pi A}}$	SI = Shape index P = Perimeter of a patch A = Area	Hole (1953, 1978)*
	Degree of chronological uniformity	Area occupied by Entisols (younger soils) and soils of other orders	-	Hole (1983)*
Size and shape	Mean size of ESA	$MS = \frac{\sum_{i=1}^{i=k} P_i}{k}$	MS = Mean ESA size P _i = Areal size of the ESA k = Number of ESAs	Fridland (1976)
	Degree of differentiation of the size of soil contours	$DD = \frac{\sum_{i=1}^{i=k} (P_i - P)}{kP}$	DD = Degree of differentiation P _i = Areal size of the ESA k = Number of ESAs	Ostrowski and Jankowki (1969)*
	Variability of the size of soil contours	$V = W/R$	V = Variability W = Number of intervals accommodating the areas of contour of the set under consideration R = Number of established intervals of the area sizes of contours	Fridland (1976)
	Coefficient of dissection	$CD = \frac{S}{3,54\sqrt{A}}$	CD = Coefficient of dissection S = ESA perimeter A = Area of the ESA	Fridland (1976)
	Boundary index	$BI = nS + mD + pG$	BI = Boundary index n, m, p = Proportion of the total ESA perimeter S, D, G = Boundary nature (sharp, distinct, gradual)	Fridland (1976)

	Nearest neighbour index	% of the length of ESA boundary occupied by the classificational groups of soils forming the neighbouring ESAs	-	Fridland (1976)
	Degree of qualitative differentiation of soils individuals	$CDSC = \frac{\sum_{i=1}^{i=n} E_i}{m} \frac{1}{n}$	<p>CDSC = Coefficient of classificational differentiation of soil cover components</p> <p>n = Number of taxonomic levels required for the description of the soil cover components</p> <p>m = Total number of soil units</p> <p>E = Number of soil units on every taxonomic level</p>	Ostrowski and Jankowki (1969)**
Size and shape	Fractionation index	$FI = \frac{k}{\sum_{i=1}^{i=k} P_i}$	<p>FI = Fractionation index</p> <p>k = Number of ESAs</p> <p>P_i = Size of ESAs</p>	Glazovskaya (1964)** Godel'man (1968)**
	Coefficient of dissection	$FI = \frac{IC(A - S_{\max})}{S * A}$	<p>FI = Fractionation (complexity index)</p> <p>IC = Mean coefficient of dissection for the ESAs on a plot</p> <p>\bar{S} = Mean size of the ESA on the plot</p> <p>S_{\max} = Size of the largest ESA on the plot</p> <p>A = Area of the plot</p>	Dimo (1953),** Godel'man (1968)**
	Entropic measure of differentiation homogeneity	$H^* = \log_2(a) - \frac{1}{a} \sum_{k=1}^n a_k \log(a_k)$	<p>H^* = The entropic measure</p> <p>a = The weight of the whole</p> <p>a_k = Weight of the n parts of a</p>	Gurevich (1968)**

*: Reference in Hole and Campbell (1985); **: Reference in Fridland (1976)

Several points should be addressed during soil landscape analysis (Hole and Campbell 1985):

- (1) *Setting*: a summary of information about a given area, including three kinds of soil cover patterns: material patterns (geomaterials, pedomaterials, biomaterials, homomaterials), form patterns (geoform, pedoform, bioform, homoform) and microclimate patterns (geo-local-climate, pedo-local-climate, bio-local-climate, homo-local-climate).
- (2) *Scale factor*: local relief classes, degree of smoothing and the size of the elementary soil bodies.
- (3) *Principal kinds of patterns on a plan view*: undifferentiated or differentiated (with subclasses such as patchwork, punctuate, point-centred, line-centred).
- (4) *Origin of the soil cover pattern*. Independently of the nature of the original landform, the specific changes taking place after the deposition (time-zero) are classified as: geologic, pedologic, local-climatic or biologic.
- (5) *Measurements for soil bodies and soil cover description*: density of soil body units per unit area, composition, soil cover diversity, etc. (Table 5.3).

4. Pedodiversity, Soilscape and Landscape Ecology

The necessity to understand the importance of landscape dynamics, heterogeneity and environmental changes has been increasingly acknowledged (Forman 1995). Even if landscape ecologists' point of view is largely biological, the soil ecosystem is relatively understudied, despite the fact that an understanding of soil ecology is essential for natural and agricultural ecosystems (Decaëns 2010, Parker 2010). In contrast, the diversity of abiotic components (e.g., soils and landforms) has received minimal attention, even though their spatial and temporal variation may produce important quantitative and qualitative changes in the landscape (Ibáñez et al. 1995). Nevertheless, interest in the abiotic portion of landscapes has its roots in physical geography, geomorphology, geology and pedology. The Russian geochemist Polynov and his colleagues focused their attention on the geochemical description of landscapes. Independently, Huggett (1975) proposed the concept of the soil-landscape system, i.e., any landscape unit in which landforms and soils, and the geomorphological and pedological processes which create them, are seen as a whole. This concept was designed to link soil processes and geomorphological processes in a landscape, a theme pursued by pedologists with a geomorphological orientation, and geomorphologists interested in soils (Huggett 1995 and references therein). Also Tricart and Kilian (1982) proposed the morphogenesis-pedogenesis balance to

understand landscape evolution, with soils playing an important role in this approach.

Pedodiversity is conceptually defined as the inventory of the variety of discrete pedological entities (e.g., soil taxa) and the analysis of their spatial and temporal patterns. Measurements of diversity were introduced to pedology few years ago (Ibáñez 1986, 1996, Ibáñez et al. 1990, 1994, 1995, McBratney 1992, 1995). McBratney (1995) proposed several approaches to pedodiversity analysis:

1. *Taxonomic pedodiversity* (the diversity of soil classes or even of diagnostic horizons). The teams led by Phillips and Ibáñez and their respective collaborators, among others, have applied mathematical tools widely used in ecological studies to the quantitative analysis of soil genesis from a new perspective (e.g., Guo et al. 2003, Ibáñez et al. 1998, 2005, Ibáñez and de Alba 2000, McBratney et al. 2000, McBratney and Minasny 2007, Phillips and Marion 2004, 2005, Scharenbroch and Bockheim 2007, Toomanian et al. 2006).

For example, richness and diversity indices, the fitting of probabilistic distribution models and richness-area relationships, were applied to the soil pattern of the Jarama-Henares interfluvium (Saldaña and Ibáñez 2004). Pedodiversity was computed considering pedons and the area covered by a given soil type, at different hierarchy levels of the USDA taxonomic classification as well as for diagnostic horizons. It is remarkable that the plots under analysis were rather homogeneous from the pedologic and geomorphologic points of view. Genetic pedorichness and pedodiversity increased with the age of geomorphologic surfaces, suggesting a divergent soil evolution. The authors suggested that the analysis of the entire solum should be considered for appropriate pedodiversity quantification. Besides, the high correlation between diversity indices applied suggested that the Shannon index, commonly used in many disciplines, might be enough to summarize all the information about diversity. Regarding richness-area relationships, the low and middle terraces fitted a logarithmic function, while the high terrace fitted a power model, in agreement with the ecological literature on biotaxa-area relationships.

Saldaña and Ibáñez (2007) analyzed the relationships between pedodiversity and spatial variability and provided another approach to the understanding of the soil heterogeneity of the area by using connectance. They found that the terrace with the highest taxonomic pedodiversity at Great Group and Subgroup levels of the Soil Taxonomy, showed a low spatial variability of soil properties and the lowest connectance values. The authors concluded that the applicability of log-normal or power-law distributions to older and

richer soil assemblages, the increase of pedodiversity over time, the suitability of the power-law model to taxa-area relationships, and the decrease of the values of connectance indices for soil assemblages are signatures that support the non-linear systems hypotheses.

2. *Functional pedodiversity* (e.g., the soil behaviour under different uses). The relationships between soil diversity and soil functions as well as the driving forces of soil biodiversity patterns are poorly understood (e.g., Decaëns 2010, Nikitin et al. 2010, Parker 2010).
3. *Diversity of soil properties*. Geostatistical tools are usually used in the analysis of continuous variables. Saldaña et al. (1998) applied statistical and geostatistical procedures to several soil properties of a chronosequence on the terraces of the Henares River. Three terraces of lower, medium and upper Pleistocene age were selected for this purpose and an intensive nested sampling scheme was applied (Fig. 5.1). They observed and quantified a decrease in variability of several soil properties from young to old deposits, showing an increment of soil homogenization with time. Geostatistics can be applied to determine the variability of single soil properties at a given depth, but this is undertaken at the expense of soil as an individual or as a soil body. Pedodiversity and spatial variability of soil properties are not synonymous concepts, because soil evolution may lead simultaneously to homogeneity and heterogeneity. This is not inconsistent because, according to systems theory, elements of a certain hierarchy level have properties that cannot be inferred from the analysis of their constituents and their variations may differ (Saldaña and Ibáñez 2007).

Soil landscape and ecological landscape patterns may be integrated. For example, Fridland (1976) introduced soilscape indices in the analysis of soil combinations; he also analyzed the spatial variation of soil properties by means of graphs and conventional statistics (e.g., coefficient of variation), and in this sense, he was a “precursor” of pedometrics. Hole and Campbell (1985) took into consideration Fridland’s ideas on soilscape and were aware of Forman and Godron’s approach. They focused on heterogeneity or diversity of soil cover pattern taken as the number of soil bodies and the degree of contrast between them, or using soil taxonomic complexity, landscape positions, and disturbance characteristics represented in an area. They also proposed metrics such as “measures of composition” which are measures of richness (Table 5.3). Later, Minasny and McBratney (2007) claimed that the taxonomic distance should be incorporated into pedodiversity calculations to obtain an effective estimate (but see Chapters 1 and 2 in this book). Therefore complexity, metrics and disturbances were already considered as a part of the quantitative analysis of the soilscape. Ibáñez (1986) studied a mountainous area in Central Spain, merging

landscape concepts into the soil cover analysis. His idea was that only with a deep knowledge of the two parts, the biotic and the abiotic, could landscape structure be understood. Saldaña (1997) applied landscape metrics to the analysis of the soil mantle in Central Spain (see section on Quantification of Landscape Pattern). Lacunarity analysis (Plotnick et al. 1993) has been applied by landscape ecologists to soils at an individual scale (e.g., solid-pore distribution pattern) but not to quantitatively characterize the soil mantle. This could be an interesting future line of research (Ibáñez et al. 2009). Hole and Campbell (1985) acknowledged that similarities exist between soil landscape and ecological landscape pattern; they found that detailed congruence of the two patterns was commonly lacking and this statement still remains unchanged.

One interesting aspect is that modern landscape ecology is based on the patch mosaic paradigm, in which landscapes are conceptualized and analyzed as mosaics of discrete patches. While this model has been widely successful, there are many situations where it is more meaningful to model landscape structure based on continuous rather than discrete spatial heterogeneity (McGarigal et al. 2009). Pedologists have some experience on this topic (e.g., Ibáñez et al. 2006, Ibáñez and Saldaña 2008).

Both, landscapes and soilscape are complex systems, and scale is a central matter in their approaches. The formation of the soil mantle generates different soil bodies and pedotaxa. Depending on the perspective and scale of analysis, soil evolution may lead simultaneously to homogenization (convergent pedogenesis) and heterogenization (divergent pedogenesis) (Phillips 2001). Applications of several mathematical tools belonging to the science of complexity and non-linear dynamics are growing in ecological and pedological literature at different scales, from small plots to planetary level (Caniego et al. 2006, 2007, Holland et al. 2009, Ibáñez et al. 1998, 2005, Irwin et al. 2009, Tang et al. 2009). Although there is still much to be done, this approach might help in detecting similarities and differences between biological and non-biological natural resources.

5. The Conservation of Pedodiversity

The Earth is changing due to rapidly increasing human populations, widespread development, and exploitation of natural resources (Amundson 2006, Wiens 2009). These modifications have led to a loss in biodiversity and changes in ecosystem services (Millennium Ecosystem Assessment 2005). The relationship between biodiversity and ecosystem functioning is a central issue in ecological and environmental sciences (Loreau et al. 2001, Maestre et al. 2012, O'Farrell et al. 2010) and has become the objective of recent conservation approaches (O'Farrell et al. 2010). The rationale behind these ecosystem service based approaches for conservation is that by

understanding and mitigating the threats posed to ecosystem services the biodiversity that underpins these services will also be conserved, while at the same time increasing the relevance, incentives and funding resources of these conservation efforts (Vira and Adams 2009).

Concern over the fate of terrestrial biotic diversity in the face of increasing human domination of the planet has focused mainly on the aboveground flora and fauna (Amundson et al. 2003). Soils have hardly been considered in this respect. Still, soils are a part of ecosystems that provide services to mankind, such as the water cycle and nutrient cycle, carbon sequestration and the maintenance of a large variety of organisms, contributing to biodiversity (Blum 2006). Many natural pedotaxa suffer a serious risk of extinction, whereas human activities have resulted in other new pedotaxa that are distributed worldwide (Ibáñez et al. 2008). Amundson et al. (2003) showed that 4.5 percent of the more than 20,000 Soil Series in the U.S. are in danger of substantial loss, or complete extinction from agriculture, industrialization and urbanization. Long-term cultivation decreased the pedodiversity in Sicily (Lo Papa et al. 2011). Soil sealing currently covers 4 percent of the total land area worldwide although more dramatic changes have occurred in some industrialized countries and several regions of the developing countries (e.g., EASAC 2009). The European Union considers urbanization to be one of the leading causes of decline in soil biodiversity in the region (Jeffery et al. 2010). Soils with minor areal extension (also termed soil minorities) are especially endangered; these rare threatened soils may be habitats for unique communities of plants or soil animals. Recently, pedologists have focused their research on soil endemism (e.g., Amundson et al. 2003, Bockheim 2005). This concept is promising for identifying rare, unique, and endangered soils (Bockheim 2005 and Chapter 8 in this book). Most of these rare soils are closely related to geodiversity and biodiversity, because typically these are soils formed in rare geologic materials (Amundson et al. 2003), or under endemic vegetation, or they result from an unusual combination of the geological and biotic characteristics of the site. Therefore, rare soils or soil minorities could be considered as soil refugia and maintained in an undisturbed state for conservation.

A landscape approach is required to study the role of soil as an integral component of natural and converted (managed) ecosystems (Dumanski et al. 2002, in Dumanski 2006). Naveh and Lieberman (1984) suggested that landscape ecology was increasingly recognized as a powerful scientific basis for land and landscape assessment, planning, management, biodiversity conservation, and reclamation. Regarding conservation, Wiens (2009) recently asserted that their integration is far from complete because landscape ecology and conservation share a common focus on places, but with different perspectives. To conservationists, the goal is to find ways to

maintain biodiversity by targeting and prioritizing places for protection or conservation management and by advocating sound environmental policies. To landscape ecologists, the goal is to use an understanding of landscape patterns and processes to design and manage land use in ways that promote the well-being of people and nature. To be effective, conservation should care about populations of species, the ecosystems of which they are a part, and the habitats they need for establishment. In many cases, this means protecting places for biodiversity. To a landscape ecologist, the habitats that so interest conservationists are elements in a larger landscape mosaic. It is the structure, spatial configuration, and context of these places, and the ways in which these influence ecological processes and undergo change, that landscape ecologists focus on (Wiens 2009). Therefore landscape ecologists, conservationists and soils scientists should find ways of collaboration. Biodiversity conservation cannot be effective without measures to conserve the diversity of soils (Khaziev 2011) and landscape ecology could be of great help.

In the case of soils, the knowledge about soil diversity and its relationships with soil functions is poor (e.g., Decaëns 2010, Parker 2010, Petersen et al. 2010). The point is what has to be protected. Ibáñez et al. (2008) proposed that there are at least three types of pedological entities that should be preserved: (i) the natural soil cover (i.e., least disturbed soils); (ii) the cultural sustainable soil cover (i.e., under farming practices which have been sustainable for centuries or millennia), and (iii) the soils of the past (Retallack 2001). Recently there has been a shift of conservation planning to consider: (1) sets of reserves that make complementary contributions to the protection of regional biodiversity, (2) the scale effect, (3) the threats from landscape surroundings to the biodiversity within the preserve, and (4) human influence and sustainability (Wiens 2009).

The design of a network of soil reserves requires measures of diversity to decide “where” and “how.” In this way the best combinations of available areas to capture the maximum diversity can be identified (Williams et al. 1996) while also considering soil minorities or endemisms and singular biological soil refugia. For conservation management, there are three approaches to preserve biological diversity (Williams et al. 1996, Williams 2008) and, by analogy, soil diversity: (a) hotspots of richness, which simply select the richest in taxa areas; (b) hotspots of range-size rarity, such as rare occurrences or rare soil assemblages (soil endemisms or soil minorities); and (c) sets of complementary areas, which select areas because of the highest combined species richness. Ibáñez et al. (2008) discussed the advantages and disadvantages of each of them.

The most popular approach in conservation biology, but probably not the optimal approach, is to select priority areas as the criterion of richness hotspots. However, according to literature, when applied to soils, the design

of a network of soil reserves would be based on the complementary areas approach. McBratney (1992) proposed a preservation strategy based on pedodiversity. Then, the most pedodiverse soil reserve candidate should be selected and others with the highest remaining total pedodiversity excluding the ones already present, always choosing the best region to compensate for groups not yet incorporated. Another example comes from Ibáñez et al. (2008) who applied complementary methods to the pedotaxa of the Aegean Archipelago. They used the cumulative soil richness to establish the minimum island size required to preserve all pedotaxa for each of the six types of island that had been differentiated based on their lithologies and soilscares. Thus complementary methods could be considered for the design of a network of soil reserves. The authors concluded that the proposed network could also be an efficient way to preserve soil characteristics and qualities of undisturbed soils that would become benchmark sites for soil monitoring programs. Interestingly, complementary area methods distinguish between irreplaceable and flexible areas, which help planners by providing alternatives for negotiation. Whilst hotspots of richness and rarity would fail to represent “all undisturbed taxa” at least once, complementary areas would represent all taxa at least two or more times in a given percentage of the study area (Williams et al. 1996). In conclusion, a network of soil reserves may be useful for preservation of soil cover and soil biodiversity conservation, for soil quality monitoring and to increase our understanding of the role of pedosphere in supplying ecosystem services.

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CHAPTER 6

Pedodiversity and Landforms

Norair Toomanian

1. Introduction

To make a general abstraction and integrate sciences involved with the biotic and abiotic phenomena in terms of their evolutionary relations, measurements of distributions, variability of properties and taxonomic diversities are required. This abstraction may enable an understanding of the way in which nature acts and how each phenomenon, in combination with the others, reacts. This also would highlight the direction of environmental co-evolutionary pathways in which these variabilities and diversities occur. From a systemic point of view, the diversity of these processes also increases when descending in more detailed relationships and induction levels (Phillips 2001a,b).

The assessment of abiotic diversity is crucial for the interpretation of the geological history of the Earth, past and present climates and landscapes, and the origin and evolution of life (Petersen 2008). It is essential for an understanding of the patterns and processes of landscapes and their biotic entities. Despite the awareness of problems such as species extinction and environmental changes in the field of biology, Earth scientists have only recently become involved in environmental conservation strategies of the abiotic components (Gray 2004).

Investigations of ecological phenomena at different spatial scales often requires quantifiable descriptions of landscape patterns and structure for testing relationships or making predictions about landscape dynamics and evolution (as discussed in more detail elsewhere in this book).

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Therefore establishing the taxonomic structures with different categories is an unavoidable activity for scientists. In natural sciences recognition of changing patterns requires subdivision of the whole object into discrete bodies such as geofoms, soil types, geologic formations, plant types, climatic zones, etc. The distribution and variation of these natural bodies will depend on their initial conditions and evolutionary pathways as well as perturbations that occur through time, resulting in spatially and temporally scale-dependent patterns and scale-invariant objects as fractals. Geologic formations, landforms and soil types are related to environmental abiotic bodies, which are formed and evolved through related geologic, geomorphic and pedologic processes over time. In geomorphology, the word “landform” is frequently used as a general term covering any type of geomorphic unit without hierarchical distinctions, but it could be considered as a type of geofom (Zinck 1988). The aims of this chapter are to (i) present the diversity structure of landscapes and soilsapes and (ii) check whether the patterns of biodiversity, geomorphological diversity and pedodiversity have similarities among different arid regions of Iran.

2. Background

2.1 Geomorphology

Geomorphology is the study of landforms and the evolution of the Earth’s surface. It attempts to explain why landscapes look as they do in terms of the structures, materials, processes and history affecting other natural bodies of the environment (Grotzinger et al. 2007). It is well known that the structural patterns of geology, hydrology, geomorphology, pedology and biology and their evolutionary processes are interrelated (Grotzinger et al. 2007). This highlights the intensity of structural interactions that existed during the formation and evolution of these naturally evolved features. The geomorphological evolution of the Earth’s surface is strongly connected to endogenous and exogenous processes; the resulting landforms are being reformed through time. Geomorphology and particularly geomorphological mapping permit us to identify and analyze landforms and to infer their origin and evolution (Pavlopoulos et al. 2009). The utility of geomorphological maps to study the soil-landform relationships, as well as correlation of their repercussions with other environmental phenomena, is well known (Pavlopoulos et al. 2009).

Soil geomorphology as a branch of the geomorphology science, records the formation and evolutionary history of Earth surface at any geomorphic position. Birkeland (1999) defined soil geomorphology as the study of soils and their use in evaluating landform evolution, age and stability, surface processes and past climates. Wysocki et al. (2000) more broadly defined

soil geomorphology as the scientific study of the origin, distribution and evolution of soils, landscapes and surficial deposits and the processes that create and alter them. Perhaps the preferred definition is that soil geomorphology is an assessment of the genetic relationships between soils and landforms (Gerrard 1992). Soils are strongly linked to the landforms upon which they develop (Schaetzl and Anderson 2005). Therefore, soil geomorphology has emerged to deal with soil-landscape relations. Both soil geomorphology and pedology share the same factors of formation and have originated from endogeneous and exogeneous energy sources. Internal geodynamics through tectonics and volcanism govern the formation of heterogeneous relief and the structural substrate on which geoform and soils are formed. Additional controls include hydrologic, erosional-depositional and pedologic processes (Farshad 2006).

2.2 Geomorphological mapping

Description, classification, and mapping are a vital task in natural resource sciences. The utility of geomorphological cartography becomes more and more imperative when scientists start to study the environmental impacts on how the Earth's surface has evolved (Pavlopoulos et al. 2009). The primary purpose of landform mapping is to show the distribution of landforms and to characterize the processes that sculpt the landforms (Paron and Vargas 2007). The concept of landform mapping or terrain classification has been developing since the early 1960s, since the easier access to aerial photography. It arises from the concept that all landscapes can be divided into smaller units (Paron and Vargas 2007). In geomorphology, unlike in geology or soil science, there is no unequivocal standard for mapping land features, or terminology, therefore different types of mapping systems, depending on the aims, materials and work scales, have been developed. The development of geomorphological mapping systems has followed different paths, partially due to the different interests and perspectives of geomorphologists and to real or perceived differences in the landforms found in various regional settings.

There are basically two philosophies in classifying soils and landforms: i) genetic and ii) morphologic classification (Schaetzl and Anderson 2005). The genetic approach is based on the genesis they had and what processes have ruled the formation of soils and landforms. The morphologic approach excludes the genetic interpretation in favor of an emphasis on measurable properties of soils and landforms. Geomorphologists have periodically used one of these two frameworks to classify soils and landforms (Smith et al. 2002). Different scientists have used different criteria for geomorphological classification, which have been changing over time (Zinck 1988, Farshad 2006, Paron and Vargas 2007). A number of attempts have been made to

develop geomorphological hierarchical classifications (taxonomy). In these taxonomies small features are nested within larger features; thus, different scaled spatial processes can be described and differentiated (Rowentree et al. 2000). Among morpho-genetic and hierarchical landform classification systems, the approach of Zinck (1988) has been called the Geopedologic approach. In this approach the morphometry (analysis of the geometry of land surface features); Morphogenesis (analysis of the genetic factors of land surface features); Morphochronology (analysis of the time factor in the creation of land surface features); Morphodynamic (analysis of the levels of activity of different features) of the landforms are considered. This approach defines hierarchical categorical levels of land classification (moving from the biggest to the smallest) which is shown in Table 6.1. The study area in most studies covers only categories smaller than morphogenetic environment; therefore, the geofoms are classified from landscape level to smaller units. Sequential criteria for differentiating geofoms are described in Table 6.2.

2.3 Soil—landform relationship

Soils are the critical interface among the atmosphere, lithosphere, biosphere and hydrosphere and are, thus an ideal integrative component reflecting a variety of interactions. During the past, pedologists and geomorphologists have understood that the relation between the landform and its processes is axiomatic (Stallins 2006). Soils are a vital resource to plant communities. In addition, soils contribute substantially to biodiversity (soil biota). The interaction between geomorphic and ecologic landscape components has been largely conceptualized without attention to soils. The diversity of soils and landforms has marked quantitative and qualitative effects on the landscape (Ibáñez et al. 1995). Both the soils and landforms, for instance determine the drainage patterns and storage of water, which, in turn, influence the structure of plant communities in a given territory (Parsons 2000).

Investigations of ecological processes at different spatial scales often require quantifiable descriptions of landscape pattern and structure for testing relations or making predictions about landscape dynamics and evolution (Saldaña et al. 2011). In order to assess landscape diversity, it is necessary to estimate the diversity of individual landscape components as the constitutive elements of landscape complexity and heterogeneity (Parsons 2000). Only through amalgamating the component diversities will an overall measure of landscape diversity be achieved (Aspinall 1996).

Table 6.1 Geoform classification system (Zinck 1988).

Level	Category	Generic	Definition
6	Order	Geo-structure	Large continental portion characterized by a broad geologic structure (e.g., cordillera, geosyncline, shield).
5	Suborder	Morphogenetic environment	Broad type of biophysical medium originated and controlled by a style of internal and/or external geodynamics (e.g., structural, depositional, erosional, etc.).
4	Group	Landscape	Large portion of land characterized either by a repetition of similar patterns or an association of dissimilar one (e.g., valley, plateau, mountain, etc.). It is mainly determined by endogenic forces such as orogenesis and volcanism. It maintains lithological and tectonic uniformity. Examples: Valley; Plain; Peneplain; Plateau; Piedmont; Hill and Mountain. In some cases is quite difficult to determine a boundary of this type of land subdivision. Average linear magnitude is of 10^0 – 10^2 km.
3	Subgroup	Relief/Molding	Relief represents morphology of Earth's surface determined by a given combination of topography and geological structure (e.g., cuesta relief-type). Molding as determined by specific morphoclimatic conditions or morphogenetic processes (e.g., glacia, terrace, delta, etc.). Average linear magnitude is of 10^0 – 10^2 km.
2	Family	Lithology/faces	Petrographic nature of hard rocks, subdivided into the three main genetic classes (sedimentary, igneous and metamorphic) and into many sub-classes. It is derived directly from existing geological maps integrated through image interpretation.
1	Subfamily	Landform	Conspicuous basic geoform type, characterized by a unique combination of process, geometry, dynamics and history. The smallest unit considered here. Corresponds to the Pedogenetic environment, regarded as the sum of topographic form, geomorphic position and geochronology unit. In most cases is a further subdivision of Relief. Dominated by exogenous processes. Average linear dimension is 10^0 – 10^{-1} km.

2.4 Pedodiversity and landforms

Spatial variability of soils has long been recognized as a crucial issue for the understanding of ecological patterns. Soils are one of the main abiotic habitat heterogeneity components reflecting the influence of many

Table 6.2 Sequential identification of geoforms (Zinck 1988).

Observation platform	Observation area	Observation features	Criteria used, Inferred factors	Resulting Geoforms	Derived generic category concepts
Satellite	Large portion of a continent	Longitudinal, narrow, high relief mass; abrupt limits	Topography, internal geodynamics (orogenic area)	Cordillera (folded mountain chain)	Geo-structure
		Large, flat, low-lying relief mass	Topography, internal geodynamics (subsiding area)	Geosyncline (sedimentary basin)	
Airplane	Cordillera	Longitudinal highlands formed of parallel mountainous chains; strongly dissected	Topography, external geodynamics (erosion)	Structural/erosional environment	Morphogenetic environment
		Sequence of flat lowland areas between chains; concave rims	Topography, tectonics, external geodynamics (deposition of sediments)	Depositional environment	
Helicopter	Structural/erosional environment	Parallel mountainous ridges	Topography, tectonics, hydrology	Mountain	Landscape
		Longitudinal, narrow depression parallel or perpendicular to ridges	Topography, tectonics, hydrology	Valley	
Earth surface	Valley	Staircase configuration, parallel topographic levels separated by scarps	Topography	Terrace	Relief/Molding
		Valley bottom, river system, riparian forest	Topography, drainage, vegetation	Flood plain	
Above and beneath the Earth surface	Terrace	Longitudinal, narrow, convex bench; well drained; coarse-textured	Topography, drainage, morphogenesis	Levee	Landform
		Large, ample, concave depression; poorly drained; fine-textured	Topography, drainage, morphogenesis	Basin	

environmental factors (Ibáñez et al. 2005a,b). The concept of diversity is fundamental in many aspects of environmental investigations (see Chapters 1 and 2 in this book). Pedodiversity can be assessed using a hierarchical methodology by classifying the landscapes into geomorphic units (Parsons 2000).

The diversity of both soils and landforms has received only a little attention in the literature (Ibáñez et al. 1990, Ibáñez et al. 1995, Ibáñez 1996). Landforms are landscape elements on which the data should be grouped and analyzed (Parsons 2000). Ibáñez et al. (1995) reported that the characterization and quantification of the diversity of landforms, geology, and soils as a non-renewable natural resource should be considered when estimating the ecological values of landscapes. This stresses the need for a testable hypothesis to be formulated for explaining, quantifying, and modeling these spatial-temporal landscape patterns. Ibáñez et al. (1990) found that patterns of plant diversity, geomorphic diversity, and pedodiversity have great similarities. This suggests that the controls on the structure and organization of biotic and abiotic components have universal similarities (Parsons 2000). Ibáñez et al. (1990) used diversity indices to describe the complexity of pedogeomorphological landscapes in a study on the hierarchical organization of drainage basins in Spain. This author applied the same methodology to analyze the evolution of fluvial dissection landforms (Ibáñez et al. 1994). Ibáñez et al. (1995) found that the patterns of biodiversity, geomorphological diversity and pedodiversity have great similarities according to species-area relationships and abundance distribution models described in the ecological literature. Ibáñez et al. (1995) emphasized the need to draw testable hypotheses for the explanation and the quantification of the underlying regularities.

3. Landform Diversity Analyses of Some Dry Land Regions in Iran

3.1 Data acquisition

The data and results of different investigations executed by the author or under his supervision are reassembled here to draw some interesting generalizations. These data sets have interesting information for different aspects of pedodiversity and soil assemblages in several regions of Iran. Descriptions of the six climatic, geomorphic regions which are undertaken in this work are showed in Table 6.3. In the rest of this chapter, the study-regions will be mentioned by their assigned capital letter (A, B, C,...). These datasets contain environmental, soil and landform information at different scales to permit the analysis of different aspects of pedodiversity, soil and landform pattern and discuss some methodological aspects. Soil

Table 6.3 Climatic and geomorphic characteristics of studied regions in Iran.

Region	A	B	C
Studied by	Toomanian et al. (2006)	Jaafari et al. 2012	Nabiolahi et al. 2012
Geology	Cretaceous different Limestone	Limestone, dolomite, shale	Limestone and igneous assemblages
Hydrologic character	Part of Zyandehrud valley	A little sub basin	Part of high hills and dissected upper terraces
Geomorphic Character	Small mountains (5%), sloping piedmonts (30%), flat playas (40%), river terraces (25%)	Sloping piedmonts (50%), Flat Playa (40%), Sand dunes (10%)	Steep hills and undulating alluviums with their interfluves
TAP, MAT, MAE ¹	110 mm, 14°C, 1590 mm	61 mm, 22°C, 1750 mm	338 mm, 10.5°C, 1200 mm
Climate (soil T and M ²)	Arid (Thermic, Aridic)	Intensively arid (Thermic, Aridic)	Semi-arid with cold winters (Mesic, Xeric)
Studied by	Moravej et al. 2012	Toomanian et al. (2012)	Esfandiarpour et al. (2009, 2010)
Geology	Marns, Conglomerates, Siltstones	Sulfidic limestone	Cretaceous limestone
Hydrologic character	Part of a big alluvial fan	Part of a bajada with some dissections	Part of a hilly and undulating alluvial bajada
Geomorphic Character	Hills (15%), Sloping alluvial fans (85%)	Sloping bajada (60%), moderately undulating alluviums (40%)	Sloping hills (50%) and undulating alluviums (50%)
TAP, MAT, MAE ²	126 mm, 19°C, 916 mm	83.9 mm, 15.8°C, 1600 mm	254 mm, 10.7°C, 645 mm
Climate (soil T and M ²)	Weak aridic (Thermic, Aridic)	Intensively arid (Thermic, Aridic)	Semi-arid with cold winters (Mezic, Xeric)

1-TAP—total annual precipitation; MAT—mean annual temperature; MAE—mean annual evaporation 2-Soil Temperature and Moisture regimes

Classification used was the USDA soil taxonomy (Soil Survey Staff 2010). The landforms and soils of each region has been formed and evolved with different geologic, hydrologic, geomorphic and pedologic history.

The influence of pedogenic processes of different regions are presented in Table 6.4. Different taxa of each region have been formed with interaction of mentioned soil forming processes incorporated with erosion-deposition of sediments with different clasts and particles sizes from different parent materials. In the regions context, topography and intensity of drainage patterns play an important role.

Table 6.4 The pedologic processes which directly differentiates soil taxa in studied regions.

Region	Processes	Landscapes of activation
A	Salinization	Playas, hills and flood plains
	Alkalization	Playas
	Gypsification	Piedmonts
	Argilification	Piedmonts (paleo), river alluvial plains
	Calcification	Piedmonts
	Aggregation (Cambids)	Piedmonts
B	Gypsification	Piedmonts, hills
	Salinization	Playas
	Calcification	Piedmonts
	Aggregation (Cambids)	Piedmonts
C	Aggregation (Inceptisols)	Catena and interfluves (hills)
	Calcification	Catena and interfluves (hills)
	Humification	Interfluves
D	Fluviation	Piedmonts
	Aggregation (Cambids)	Piedmonts
	Gypsification	Hills
E	Calcification	Piedmonts
	Gypsification	Piedmonts
	Salinization	Piedmonts
F	Argilification	Piedmonts
	Aggregation (Inceptisols)	Hills and piedmonts
	Calcification	Hills and piedmonts

3.2 Geomorphologic stratification

Geomorphology defines the main differentiating resources (land components) and pedology defines detailed differentiating factors of the soil map units (soil components) (Saldaña et al. 2011).

A geopedologic unit is more or less equivalent to a soilscape unit. All the regions were delineated by air photo interpretation (1:50,000 or

1:20,000) depending the size of the study area. It means that 1:50,000 and 1:20,000 scaled air photos were used to execute the aforementioned studies. Geomorphic units were defined and delineated in all regions using a hierarchic classification system of geofoms (Tables 6.1 and 6.2) comprising four categorical levels (landscape, relief/molding, lithology and landform) as defined by Zinck (1988). All map units were described with their proper categorical legend and descriptions. The geofom legends are combined with their respective soil components. Soil components are defined by soil taxa identified in each geomorphic map unit. Using polygon maps, taxa richness as well as the abundance of soil and landforms permits us to carry on matrices to pedodiversity and pattern analyses at different levels.

3.3 Pedodiversity area relationship

In ecology, the relation between the number of species (S) and the extent of area (A) in which these species are spread, is described by power law models (Ibáñez et al. 2005b, Ibáñez and Effland 2011, Feoli et al. in this book). The total number of species is proportional to the size of the area. Strong similarities between biological and pedological assemblages have been detected based on the application of pedodiversity tools to soil maps at different scales (Ibáñez et al. 2006, 2009). Similar results also occur in field plot studies (Petersen et al. 2010). Geodiversity (geomorphic and lithological diversity) and pedodiversity together have strong influences on the architecture of ecosystem (Ibáñez et al. 2005a). The theoretical core of pedogeography and biogeography is that the number of species and soil types pedotaxa increases with area according to a power law (Ibáñez et al. 2005a, Ibáñez and Effland 2011). In this framework, the distribution patterns of landforms in different environment and their relations with the area are studied. Pedorichness of geofoms studied in the six regions are shown in Table 6.5. Pedorichness and diversity are dependent of the study scale, sampling intensity and area (see Ibáñez et al. in this book). As depicted in Table 6.5, the richness and diversity values of geofoms are in agreement with the patterns detected by Ibáñez and Effland (2011). The geopedologic diversity indices increase as the size of region increases. As Table 6.5 shows the decreasing rate of landform diversity related to area breaks when the sampling intensity goes higher than one and half point in each square kilometer. The scale and thus sampling intensity of region F is much higher than the other regions. Therefore, its diversity indices tend not to follow the decreasing rate of the studied areas.

Power-law fits of area with geomorphic and taxonomic richness as well as diversity values, are provided in Tables 6.6, 6.7, 6.8 and 6.9. The results depicted in these tables show a consistently good statistical fitness. The z exponent calculated here (0.20–0.52 for landform richness-area and 0.14–0.32

Table 6.5 Diversity indices of categorical geofoms in studied regions.

Richness index							
Region ¹	Area (Km ²)	No. of studied points	Sampling Intensity (per Km ²)	Landscape	Relief/ Molding	Lithology	Landform
A	3000	191	0.064	7	20	29	46
B	900	126	0.14	5	8	11	18
C	200	83	0.415	2	8	19	26
D	180	72	0.4	2	7	NA	28
E	80	122	1.5	2	2	3	6
F	11	94	8.5	2	2	5	11
Shannon diversity index							
A	3000	191	0.064	1.33	2.23	2.65	3.156
B	900	126	0.14	1.11	1.73	1.96	2.62
C	200	83	0.415	0.155	1.73	1.97	2.33
D	180	72	0.4	0.55	1.01	1.23	2.65
E	80	122	1.5	0.083	0.083	0.57	1.44
F	11	94	8.5	0.68	0.68	1.41	2.06

¹Scale of studies are; reconnaissance for A and B, Semi-detailed for C and D, detailed for E and super-detailed for F region.

Table 6.6 Power law fits of area with landform and soil taxa richness in regions.

	c	z	R ²	Pedotaxa-area relationship	c	z	R ²
Area-landscape	0.28	0.4	92	Area-suborder	2.03	0.14	85
Area-Relief	0.613	0.42	81	Area-great group	1.54	0.23	79
Area-Lithology	0.41	0.52	61	Area-sub great group	1.69	0.27	78
Area-Landform	4.46	0.2	65	Area-Family	4.16	0.32	75

c and z correspond to the constant and z exponent showed by Feoli et al. in this book. The region F is excluded from the calculations.

Table 6.7 Power law fits of area with landform richness in landscapes of regions.

	c	z	R ²		c	z	R ²
Region A	0.419	0.455	67	Region B	0.118	0.66	90

c and z correspond to the constant and z exponent showed by Feoli et al. in this book.

for soil taxa richness-area relation) is in range with what Ibáñez and Effland (2011) had found for island area relationship and is not far from 0.25 which is obtained for ecological studies. Williamson (1981) has also found a power law fitness between richness and area of the geological types.

Table 6.8 Power law fits of area with soil taxa richness in landforms of regions.

	c	z	R²		c	z	R²
Region A	0.65	0.37	56	Region B	0.49	0.62	76
Region C	0.91	0.64	76	Region E	1.72	0.42	90

c and z correspond to the constant and z exponent showed by Feoli et al. in this book.

Table 6.9 Power law fits of landforms area-age correlations.

Pedorichness			
Region	c	z	R²
A	1.23	0.55	78
E	0.91	0.37	65

The number of landforms in different landscapes of regions A and B (having more landforms in each landscape) also fits the power law (Table 6.7). This means that with extending the area of landscapes the landforms richness increase to higher values.

Richness of soil taxa within different landforms of regions also shows the same area-relationship phenomena (Table 6.8). Larger landforms in each region have more soil taxa richness values.

Table 6.9 shows that the power law also fits the age dependent landforms in A and E regions (landscape evolution were not studied in other regions). This confirms that the pedorichness is also positively related to age of landforms having more time to develop and diversify.

As axiom pedorichness values increase, accordingly they are applied from the top to the bottom of taxonomic hierarchy (Table 6.10). However the same does not necessarily occur with the values for evenness and Shannon indices. The diversity indices calculated within the pedological hierarchy of USDA *Soil Taxonomy* increase progressively with larger areas and lower taxonomic categories. Area is positively correlated with pedodiversity in the lower categories, but the same does not occur with the highest ones, as was also detected by Ibáñez and Efland (2011) in the Hawaii islands using the same soil taxonomy. This means that different climates and biocenoses have lesser influence on formation and evolution of landscapes and soils than local hydrologic and pedologic processes in arid regions. The main causes of soil diversity and variability and the pedogenetic evolution process are described by Phillips in this book. Landform diversity and pedodiversity follow the same patterns in the studied regions (Tables 6.5 and 6.10). This is supported by correlation matrix measured between the richnesses of geofom and soil types within geopedologic and soil taxonomic hierarchies (Table 6.11). This table shows that correlation between landform richness increases from the top to the bottom of taxonomic hierarchy and correlation between soil family richness increases from the

Table 6.10 Diversity indices of taxonomic categories in studied regions.

Pedorichness index								
Regions	Area (Km ²)	Profiles	Sampling Intensity (per Km ²)	Order	Suborder	Great group	Subgreat group	family
A	3000	191	0.064	2	6	11	16	53
B	900	126	0.14	2	6	7	12	38
C	200	83	0.41	3	4	6	8	36
D	180	72	0.4	2	4	4	5	NA
E	80	122	1.5	2	4	5	7	13
Shannon diversity index								
A	3000	191	0.064	0.51	1.68	2.036	2.37	3.47
B	900	126	0.14	0.332	1.623	1.759	2.357	3.405
C	200	83	0.41	0.554	0.582	1.04	1.19	3.34
D	180	72	0.4	0.62	1.17	1.17	1.552	NA
E	80	122	1.5	0.1443	1.17	1.325	1.65	1.98
Evenness index								
A	3000	191	0.064	0.74	0.93	0.84	0.85	0.87
B	900	126	0.14	0.479	0.9	0.904	0.948	0.936
C	200	83	0.41	0.5	0.42	0.58	0.57	0.93
D	180	72	0.4	0.903	0.844	0.844	0.964	NA
E	80	122	1.5	0.21	0.84	0.82	0.848	0.772

top to the bottom of geopedologic hierarchy. Table 6.11 confirms that the geopedologic categories are significantly correlated with the greater group and lower categories of soil taxonomy. Pedogenetic processes may act as a driving force to exaggerate soil variations associated with environmental controls, or to reduce such variations. The amplification (or filtering) of initial variations and disturbances can increase (or decrease) pedodiversity. Richness-area analysis, which is directly related to pedodiversity, is an approach to assess the dynamical instability and chaos in soils (Phillips in this book) when these are applied at detailed scales and a high sampling efforts (Phillips 2001a,b).

In studies at medium and small scales as well as with constant sampling intensity, Ibáñez et al. (2005b), Ibáñez and Effland (2011) found the same results. The results of this study support the idea that pedorichness-area relationship is more effective than sampling intensity when it is not extremely different (Tables 6.5 and 6.10).

The resulting richness-area relations suggest a similarity of pattern and structure with biological studies. Of course in a super-detailed scale of study, the relation is obscured by the high sampling intensity (e.g., region F). Table 6.10 shows that pedorichness and pedodiversity of taxonomic categories are greater with the increasing area of the regions (except region F).

Area and relief relations are the major driving forces with regards to the diverse geopedologic and pedologic taxa. In addition to these two factors, larger areas have a higher number of hydrologic, geomorphic and pedologic processes in dry lands of central Iran than smaller ones.

Table 6.11 Correlation matrix between pedorichness of geopedologic and soil taxonomic categories.

	Area	Landscape	R/M	Lithology	Landform	Order	Suborder	G. group	Subgroup	Family
Area	1									
Landscape	0.95**	1								
R/M ¹	0.94**	0.82*	1							
Lithology	0.83	0.72	0.95*	1						
Landform	0.81	0.67	0.95**	0.99**	1					
Order	-0.42	-0.48	-0.33	-0.12	-0.22	1				
Suborder	0.79	0.91*	0.76	0.61	0.57	-0.63	1			
G. group	0.97**	0.94**	0.90*	0.89*	0.73	-0.26	0.78	1		
Subgroup	0.93**	0.97**	0.84*	0.79	0.63	-0.32	0.86*	0.98**	1	
Family	0.84	0.84	0.93*	0.93*	0.940*	-0.17	0.73	0.90*	0.87*	1

*Correlation is significant at the 0.05 level; ** Correlation is significant at the 0.01 level. 1- Relief/Molding (R/M)

As it is clear from Table 6.11, there are generally significant correlations between different levels of geomorphological, pedological hierarchies and lithological variables, as well as with the area. These similarities of pedologic and geomorphic patterns mirror those of Ibáñez et al. 1990, 2005b, Phillips 1999, Ibáñez and Effland 2011, Toomanian et al. 2006, and Williamson 1981.

3.4 Effect of scale and sampling intensity on pedodiversity

Three points should be considered in scale dependent environmental analysis: Sampling intensity (scale), Small unit effect, and Nestedness. Scale is regarded as a complex concept having multiple connotations reflected in the majority of environmental studies (Martin et al. 2005). First, it is the central concept for describing and explaining the complex hierarchical organization of pedological studies (Marceau 1999). Second, all environmental entities, processes and variations cannot be studied at a single scale of observation. Therefore, there may be new findings in intensive studies compared with lower scale studies. Gao (1997) investigated the effects of sampling intensity on the accuracy of digital elevation models (DEMs) and derived topographic attributes. Their results show that accuracy tends to decrease with increasing sampling intervals, and that greater accuracy seems to be associated with more complex terrains. Also the magnitude of complexity changes more substantially at shorter sampling intervals than at longer ones. This implies that the choice of sampling interval may have a substantial effect on soil and landform complexity values; hence, the shorter the intervals, the greater the pedodiversity. This was shown by Esfandiarpour et al. (2009) when studying the effect of sampling intensity on pedodiversity of a selected landform in region F. They sampled the soils of a landform in 500, 250 and 125 meter intervals to find the most accurate distribution of soil patterns in a selected region. They found that at intense sampling intervals there were more new soil families in addition to families introduced in lower sampling intervals.

3.5 Landscape patterns analyses

To characterize the landscape heterogeneity of regions, different landscape indices (pattern metrics) were used. A brief description of these landscape indices, and a complete list of references for full definitions, is presented in Saldaña et al. (2011, as well as Saldaña's chapter in this book). The indices which are used in this investigation and their descriptions are presented in Table 6.12.

Table 6.12 Indices of landscape distribution patterns.

Type of index	Index	Carried out in	Formula	Variables
Heterogeneity Indices	Mean density of soil or landform bodies	Region	$M_d = N_d / A$	<i>Md</i> : mean density of soil bodies (soil types/area); <i>Nd</i> : number of soil or landform types; <i>A</i> : total area (km ²).
	Index of heterogeneity	Region	$HI = M_d * n$	<i>HI</i> : index of heterogeneity (landform types*map unit/km ²); <i>Md</i> : mean density of landform bodies; <i>n</i> : number of landform map units present in the area
Diversity Indices	Richness	Region	-	-
	Geopedologic diversity Index	Region	$H' = - \sum_{n=1}^m P_i * Ln (P_i)$	<i>H'</i> : diversity index; <i>pi</i> : number of delineations belonging to a given map unit; <i>m</i> : number of geopedologic units

3.5.1 Heterogeneity indices

The mean density of soil and landform bodies, *Md*, is a possible indicator of soil and landform heterogeneity (Saldaña et al. 2011). Values vary depending on the relief, climate, vegetation cover, dissection and erosion processes. These extrinsic factors vary depending the structures, environments and idiosyncratic histories of each region. The index of heterogeneity, *HI* (Saldaña et al. 2011) considers the number of the different geopedologic units present in the maps and thus is a measure of landform heterogeneity. *HI* is related to the diversity of the soil cover so that a highly diverse soilscape will give high *HI* values. The application of pattern indices to the geopedologic combinations of landforms in the tested regions are shown in Table 6.13. According to this table, the whole heterogeneity indices show no clear trend among the regions.

3.5.2 Diversity indices

The richness of soil families and landforms in regions with geopedologic diversity indices are understood as the structural heterogeneity of the studied regions. The highest diversity values corresponded to region A with the largest number of soil and landform units (Toomanian et al. 2006) and the smallest diversity belongs to region E with equitable distribution and no replication of landform types (Table 6.13). Decreasing of geopedologic diversity from a larger to smaller region is the result of

Table 6.13 Pattern indices applied to the soilscape in different regions at two levels of the geoform classification system by Zinck (1988).

Type of index	Index	Regions				
		A	B	C	D	E
Heterogeneity Indices	Mean density of soil bodies (families)	0.177	0.042	0.18	NA	0.1625
	Mean density of landforms	0.153	0.02	0.13	0.155	0.075
	Index of heterogeneity	30.6	1.42	26.91	4.65	0.45
Diversity Indices	Richness (families, landforms)	53, 46	38, 18	36, 26	NA, 28	13, 6
	Geopedologic diversity Index	3.047	2.603	2.44	3.29	0

their respective sizes and relatively acting hydrologic, geomorphic and pedologic processes in those areas. Geopedologic diversity of region D is an exception to this trend, because this study area is on an active large fan with a full range of drainage dissections. This intensity of dissection counteracts the normal trend of soil evolution comparing with the other regions.

4. Concluding Remarks

We tested the hypothesis that soil and landform taxa distributions on different regions of Iran can be modeled as a power function similar to those proposed for species–area relationship of ecosystems. As the results show, the distribution of soil and landform taxa within the structure of the studied regions are representative of a power law distribution as Ibáñez and Effland (2011) proposed for islands. Dissimilarity of parameters of power law fits (Tables 6.6, 6.7, 6.8 and 6.9) is due to different evolutionary pathways by which the landform and soils have been evolved in the different regions. Therefore, pedodiversity and geomorphological indices seem to follow similar patterns, as well as those previously established for biodiversity analysis. As abiotic evolutionary sub-systems, the diversity statistics of soil and landform patterns tend to increase with time and area, showing an increase in complexity of the system's structure. In fact the patterns of biodiversity, geomorphological diversity and pedodiversity have great similarities. This suggests that there are universal regularities common to organization of biotic and abiotic ecological structures. The quantitative analysis of the geopedologic patterns of different regions strongly support the hypothesis carried on by Ibáñez et al. (1990, 1994), Saldaña et al. (2011) and Phillips (1999). To analyze pedodiversity and relate it with biotic ecosystems, it is better to run the investigations in a naturally evolved ecological area. In such systems the natural processes are fully charged and not restricted by human abstractions.

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CHAPTER 7

Repercussion of Anthropogenic Landscape Changes on Pedodiversity and Preservation of the Pedological Heritage

Giuseppe Lo Papa^{a,*} and Carmelo Dazzi^b

1. Introduction

Over a period of time people have lived in and with their surrounding landscapes and for several thousand years transformed the soils and the vegetation into cultural landscape types important for their economy and to meet their needs (Richter 2007, Ellis 2011, Hjelle 2012). The sustainable provision of goods and services depends critically on managing soils without damaging the natural soils and the related natural resources. To support the transition towards sustainable development, science needs to understand how land-use change affects the environment and how this, in turn, feeds back into human livelihood strategies or influences the vulnerability of the environment (Rounsevell et al. 2012a). Interactions between decision-making, governance structures, production and consumption, technology, ecosystem services and global environmental change influence human activities at the local and regional

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scale, and are influenced by and feed back to the global scale, thereby shaping trajectories of human–environment interaction in land systems (Lambin and Meyfroidt 2011).

These days, over 21 percent of the world's population is living on cultivated land constituting only 7 percent of the total area available on the Earth (Jing Wang 2012) and, according to FAO estimation (FAO 2012) each person living on Earth, has at its disposal only 2400 m² to satisfy all his needs. Such enormous pressure of man on soils, have led to a striking transformation of agricultural ecosystems, to huge land-use/land-cover changes all over the world (Cocca et al. 2012, Gellrich and Zimmermann 2007, García-Martínez et al. 2009, 2010) and, many times to the creation of anthropogenically modified soils. These last can be found in several landscapes anthropogenically changed: urban areas (Short et al. 1986a,b, Agarkova et al. 1991, Burghardt 1994a,b), mining areas (Ciolkosz et al. 1985, Indorante et al. 1992, Haering et al. 2005), iron and steel production areas (Buondonno et al. 1998), agricultural areas (Dazzi and Monteleone 1999, 2007), forested areas (Andres-Abellan et al. 2005), and recreational areas (Arroyo and Iturrondobeitia 2006).

Several surveys carried out in many parts of the world on the anthropogenic influence on soils, have demonstrated that Land Use and Cover Change (LUCC) should be regarded as a primary source of global environmental change (Millennium Ecosystem Assessment 2005), and understanding its causes and consequences is one of the major goals of global change research (Lambin and Geist 2006, Lambin et al. 2003, Rindfuss et al. 2004). LUCC is driven by the interaction of ecological, geographical, economic, and social factors (Zang and Huang 2006, Sirami et al. 2010) that determine the trajectories of landscape development (Hersperger and Burgi 2009, Lo Papa et al. 2011, Dazzi and Monteleone 2007, Fialkowski and Bitner 2008, Gasparri and Grau 2009, Marin 2011). Land-use change and changes in soil management, often occur together (Halvorson et al. 2000), resulting in changes in soil quality, including soil physical features (Wang et al. 2006), soil biology (Kennedy and Papendick 1995) and soil nutrient contents (Kong et al. 2006). Modification at soilscape levels can lead also to an increase in soil aggregate breakdown, in soil organic matter losses and in soil erosion rates (Brandt and Thornes 1996, Drake and Vafeidis 2004) and to a diminishing of pedodiversity due to the consequent creation of human-disturbed soils.

Nandy et al. (2011) have shown that forest degradation by human activities such as livestock grazing and tree harvesting is related to changes in spatial patterns, while Abdullah and Nakagoshi (2008) and Zomeni et al. (2008), have revealed how deforestation associated with agricultural expansion and forest regrowth associated with land abandonment are closely linked to changes in landscape patterns. Similar findings were

obtained by Cocca et al. (2012) who estimated the changes of agricultural areas in the eastern Italian Alps, integrating socioeconomic indicators with topographical features to evaluate the drivers of such changes. Other aspects concern the interactions that occur between LUCC and biodiversity. Almeida et al. (2011) showed that in Brazil, the conversion of native pastures to exotic pastures leads to a loss of dung beetle richness and increasing dominance and changes in species composition highlighting the importance of maintaining native pastures in the Cerrado agro-pastoral landscape. Ogutu et al. (2012) demonstrated how land-use change and human population growth affected wildlife population dynamics inside Kenya's Lake Nakuru National Park during 1970–2011. In the same way anthropogenic disturbance of a tropical forest landscape in the region of the Panama Canal, impacts abundance of the triatomine bug *Rhodnius pallescens*, a vector of Chagas disease (Gottdenker et al. 2011).

In recent years and in the context of the global environmental changes, attention has focused on the relation between landscape spatial patterns (spatial arrangement and composition of soilscape elements) and human processes (large scale farming, forest fragmentation and deforestation by land-use/land-cover change) (Ferrier and Drielsma 2010, Holmes et al. 2010, Echeverría et al. 2012, Lo Papa et al. 2011, Dazzi and Monteleone 2007).

Such findings together with the many papers and reports promoted by the scientific communities under the International Geosphere Biosphere Program (IGBP 2012) and the International Human Dimension Program (IHDP 2012) testify that important progress has been made over the last few decades in understanding the land system (Rounsevell et al. 2012a). Research conducted in the LUCC program has demonstrated the pivotal role of land change in the Earth system and helped the international research community to greatly increase its understanding of the dynamics of land-use change and its consequences (Rounsevell et al. 2012b). Notwithstanding, despite recent progress in understanding the human processes that influence landscape systems, a number of great challenges remain for land-system science. Land-system science refers to the "... *interdisciplinary field [that] seeks to understand the dynamics of land cover and land use as a coupled human–environment system to address theory, concepts, models, and applications relevant to environmental and societal problems, including the intersection of the two*" (Turner et al. 2007).

To examine the repercussion of anthropogenic landscape changes on pedodiversity and the way in which such changes can influence the preservation of the pedological heritage, we propose a case study in which we: i) analyze the spatial and temporal land cover and land-use change in the period from 1955 and 2008; ii) analyze the relationships between land use and anthropogenic soils formation highlighting the influence of specific anthropogenic processes on soil evolution and stressing the importance

of a practical soil classification for anthropogenic soil management; iii) evaluate the spatial and temporal distribution of anthropogenic soils and forecast, with reference to 2050, the space-time change of the soilscape through a specific simulation model; iv) verify the soilscape diversity using several indices; and v) investigate the three-dimensional spatial variability of selected physico-chemical properties of Anthrosols, highlighting how humans can modify the natural configuration of the soils creating an unpredictable chaos of the features in the soils.

2. Study Area

The study area is represented by the Mazzarrone administrative area, a small town, located in the south-eastern area of Sicily, Italy (Fig. 7.1). In this area, characterized by a typical Mediterranean climate, landscape is quite poor and rough (maximum difference in height 200 meters), and parent materials date back to the Pleistocene and Holocene, showing a genesis that is both continental and marine (clay and sandy-clays; fossiliferous yellowish sandstones; fine quartzite sands with intercalations of well-cemented arenaceous levels; poorly cemented sands; lacustrine deposits; old and recent alluvium).

In the study area, landscape evolution and the elements that characterize it, particularly land use and soils, have to be considered on the basis of a comprehensive analysis of the economic and social factors that have influenced such an environment, considering that during the 1970's, vineyards spread copiously and produced a large increase in capital income and that led to an almost complete disappearance of unemployment. Consequently, human action should be considered as a pedogenetic factor (Dudal 2004) that has directly modified and altered the landscape and the soils in the social and economic context of the surveyed area.

The available aerial photographs and details from the Potential Vegetation Map of Sicily (Gentile 1968), indicate that most of the study area was once covered by oak and maquis (Cork oak—*Quercus suber*; locust tree—*Ceratonia siliqua*; Pistachio—*Pistacia lentiscus*; bay tree—*Laurus nobilis*; Oleaster—*Olea europaea*; Alatern—*Rhamnus alaternus* and Turpentine tree—*Pistacia terebinthus*). Statistical data and aerial photographs show that in the 1960's, together with maquis, land was mostly used for arable farming, olive and almond groves that were grown on the soils that once formed the Mazzarrone's soilscape. Such soils were represented by five Soil Taxonomy orders: Entisols, Inceptisols, Vertisols, Alfisols and Mollisols, that, subdivided in six suborders, eight great groups and 15 subgroups (Table 7.1), testify to the wide pedodiversity of the Mazzarrone area, considering that the whole Sicilian soilscape is characterized by the presence of six soil Orders (the 6th represented by Andisols).

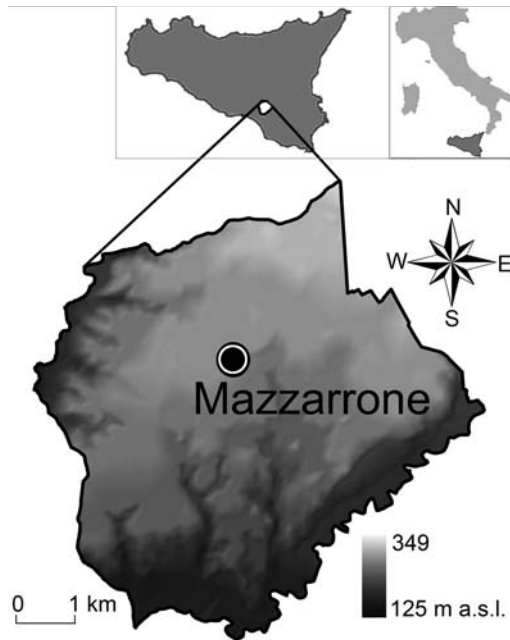


Figure 7.1 Location of the study area.

Statistical data and aerial photographs show that in the 1960's, land was used mostly for arable farming and olive groves. Vine-growing spread rapidly during the 70's, but the economic explosion occurred during the 80's, when cultivation started to be converted everywhere (Lo Verde 1995). In that period vineyards replaced arable land, almond-yards, olive groves and natural grazing at a rate that was not easily measurable and with a consistent and evident transformation of the landscape. Such large-scale farming was achieved through very deep plowing, excavations, land leveling and trenching. In many cases large amounts of "white earthy materials" were spread over the soils by trucks in order to improve the quality of the grapes. Today most of the soils that originally made up the soilscape of Mazzarrone area, have totally disappeared and have been replaced by soils that can be considered as anthropogenic (Dazzi and Monteleone 2007, Dazzi et al. 2009, Lo Papa et al. 2011). Results from unpublished soil surveys carried out in 1964 and in 1984, from aerial photo interpretation (of 1955, 1966, 1987, 1997), and the evaluation of the environmental features of the area which can be regarded as pedogenetic factors and by those of areas very close to Mazzarrone which still retain "natural" soils, allowed us to define the soils that characterized the Mazzarrone's soilscape before man's intervention in changing land use for large scale farming. In particular, on the more stable surfaces (with flat or very gentle slope), soils were made up of Inceptic,

Table 7.1 Area covered by the soils.

Order	Subgroup		
		ha	%
Entisols			
	Lithic Xerorthents	13.9	0.4
	Typic Xerorthents	541.7	15.7
	Vertic Xerofluents	71.1	2.1
Inceptisols			
	Typic Calcixerepts	192.0	5.6
	Typic Haploxerepts	979.6	28.3
	Vertic Haploxerepts	97.7	2.8
Vertisols			
	Typic Calcixererts	26.8	0.8
	Typic Haploxererts	132.0	3.8
Alfisols			
	Typic Haploxeralfs	144.9	4.2
	Inceptic Haploxeralfs	120.2	3.5
	Mollic Haploxeralfs	10.9	0.3
Mollisols			
	Typic Haploxerolls	635.5	18.4
	Calcic Haploxerolls	41.6	1.2
	Entic Haploxerolls	267.9	7.7
	Pachic Haploxerolls	129.5	3.7
Urban areas		51.7	1.5
Total		3,457	100.0

Mollic and Typic Haploxeralfs. These were moderately deep soils, with a rather shallow and not very thick argillic horizon that, depending on the morphology and mainly on the plant cover, might be overlaid by a mollic epipedon. The less stable surfaces (which were on gentle or moderately steep slopes), showed Vertic and Typic Haploxerepts, Typic Calcixerepts and Calcic, Entic, Pachic and Typic Haploxerolls. These were soils in general moderately deep with a cambic or a calcic horizon overlaid, in several cases, by a mollic epipedon. The steeper surfaces and the slope side of the stream valleys were, and in some cases still are, characterized by Lithic and Typic Xerorthents, more or less shallow soils exposed to erosion. The bottom valleys till today are characterized by Vertic Xerofluents (deep soils strongly influenced by the features of the substratum) and by Typic

Haploxererts and Typic Calcixererts (very deep and clayey soils). Today most of the Mazzarrone soilscape is made by anthropogenic soils which, according to recent proposals could be classified as Miscic Geofragmexerant (following Soil Taxonomy—Dazzi and Monteleone 2007) or Geomiscic Anthrosols (following WRB—Dazzi et al. 2009).

3. Spatial and Temporal Land Cover and Land-use Changes

Remote sensing seems to be the most useful tool to detect quickly and at a relatively low cost the transformations of the main landscape elements by means of multi-temporal analysis. In particular, with remote sensing image datasets, it is possible to evaluate the trends of land-use changes and use them to outline adaptive management strategies (Bailly and Nowell 1996, O'Regan 1996). Determination of this kind of criterion is essential for sustainable management of the environment, for assessing land carrying capacity and for avoiding irreversible degradation caused by misuse or abuse of the natural resources (van Mansvelt and van der Lubbe 1999).

Available datasets of images for the study area were panchromatic and colored aerial photographs for the following years: 1966, 1987, 1997, 2000 and 2004. The average scale of the different sets ranges from 1:33,000 to 1:7,000. Aerial photographs were acquired in digital mode at high resolution (1,200 dpi) by means of a plane scanner. Successively they were geo-referenced and ortho-rectified using a digital elevation model with a spatial resolution of 1 m and using a minimum of 60 ground control points, for every photograph, extrapolated from the most recent (2004) digital topographic map at scale 1:10,000.

Geometric re-sampling of images was carried out using the nearest-neighbor algorithm. Aerial photographs geometrically corrected were mosaic and balanced. The result was a set of aerial ortho-photo maps with a spatial resolution of 0.5 meters for each considered year. Photo-interpretation was carried out using standard photographic keys (tone, texture, pattern, shape and size). The minimum mapping unit was set at 0.5 mm for both radius of circular features and side of rectangular features. This size corresponded to real features of 5 m on topographic base map. Digital stereo-viewing of aerial photographs also was a useful support to improve the interpretation.

Using GIS software, we created land-cover maps in topological vector polygonal model for the five available years. We defined 20 land-cover units according to the CORINE Land Cover 2000 legend structure (Bossard et al. 2000) to the fourth level. Additional field inventories in 2006 were made to verify in field the correspondence of the interpreted land cover and to control the quality of the interpretation. To minimize possible interpretation errors, the work was carried out by the same person after some reconnaissance

surveys in the study area, to obtain a general understanding of the land cover situation. Land-cover units were grouped in nine land-use classes (Table 7.2). During some field surveys, we established the relation between land cover (biophysical attributes of the Earth surface) and land use (anthropic utilization applied to these attributes). Furthermore, we defined land-use classes according to grades of human influence as a soil-forming factor. Therefore, following this criterion and considering the real agricultural practices (use of fertilizers, tillage systems, the influence on erosion risks and

Table 7.2 Correspondence between interpreted land cover units and assigned land use classes for the study area of Mazzarrone. Code was used in relational database of the GIS.

Land cover		Land use	
Code ^{a)}	Unit ^{a)}	Code ^{b)}	Class ^{b)}
111	Continuous urban fabrics	I	Urban and rural fabrics and road networks
112	Discontinuous urban fabrics		
113	Rural, industrial and commercial fabrics		
122	Road networks and associated infrastructures		
21	Arable lands	II	Arable lands
25	Pastures	III	Pastures
2212	Espalier or sapling system vineyards	IV	Espalier or sapling system vineyards
134	Transformed land	V	Transformed areas and trellis system vineyards
2211	Trellis system vineyards		
222	Fruit trees	VI	Olive groves and fruit trees
2231	Olive groves		
2232	Association of olive and other groves		
224	Association of olive and vines		
244	Land principally occupied by agriculture with significant areas of natural vegetation.	VII	Heterogeneous agricultural areas
311	Broad-leaved forest	VIII	Woodlands and semi natural areas
3121	Coniferous forest with pines and cypresses in prevalence		
323	Mediterranean maquis, shrubland, garrigues		
332	Bare rock		
333	Sparsely vegetated areas		
512	Water body	IX	Reservoirs and artificial lakes

^{a)}following the CORINE Land Cover legend 2000 till to fourth level.

^{b)}arbitrarily assigned.

any other factor affecting soils), we ordered land-use classes in a sequence according to the increasing human influence on soil:

Woodlands and seminatural areas < *Pastures* < *Heterogeneous agricultural areas* < *Arable lands* < *Olive groves and fruit trees* < *Espalier or sapling system vineyards* < *Transformed areas and trellis system vineyards* < *Reservoirs and artificial lakes* < *Urban and rural fabrics and road networks*.

In the first class (*Woodlands and semi natural areas*) there are shrub-lands with typical Mediterranean species and secondarily artificial woodlands of Eucalyptus. They occur mainly in steep areas and close to the rivers course. In these areas the human impact is null or negligible.

Pastures indicate lands with herbaceous vegetation often with a rare presence of trees sparsely or cluster distributed. Vegetation is represented by grass and/or legume, trees are often typical species of the Mediterranean maquis (holm-oak, lentisk, carob). The only element of impact, in these areas, is the occasional grazing.

Heterogeneous agricultural areas class includes mainly lands with mixed natural vegetation and fruit trees (almond, olive). The low human impact is due to some occasional localized soil tillage in correspondence with cultivated trees.

Arable lands represent areas with cereals (durum wheat, barley) or forage crops (clover, vetch). The adopted agricultural systems, consist typically in one plowing per year, some harrowing and low inputs of fertilizers and herbicides.

Olive groves and fruit trees (almond, peach, plum, pear, apricot) indicate cultivated areas, except in rare cases, extensive cultivation. The cultivation techniques require deep plowing during the implant and one or more harrowing during the spring season.

Espalier or sapling system vineyards are specialized cultivation for wine production. The tillage system is the same of the previous land-use class but differs by higher inputs of fertilizers and for the use of pesticides.

Transformed areas and trellis system vineyards class include areas specialized for the table grape production.

The cultivation system involves high inputs and remarkable soil alteration. In fact, soil is covered with a layer of variable thickness of marly limestone then mixed with a mold-board, single-furrow plow and tilled to around 100 cm in depth. Vines are covered with plastic films to increase the quality of the fruits and to obtain production also during the colder season. Moreover, this is the only land use that requires irrigation and allows the

highest inputs of fertilizers, pesticides and herbicides. Transformed areas are lands created purposely for these vineyard systems and temporarily without vine plants. Recognition by photo-interpretation of these areas is quite obvious because of the presence of plastic covers on vineyards and because of the characteristic light color of the soil surface in the transformed areas, due to the addition of marly limestone layer and also due to the reshaped morphology of the landscape.

The evaluation of the spatio-temporal land-use change patterns was carried out by multi-temporal comparison of five land-use maps (years: 1966, 1987, 1997, 2000 and 2008). Areas that change and the relative direction of changes were identified using the information of the overlaid maps, obtaining four transition maps. After the topological overlay in GIS, the change detection was reclassified using a single transition matrix. In particular, we defined seven transition classes (*Abandonment*; *Extensification*; *No change*; *Persistence*; *Intensification*; *Anthropization*; *Urbanization*), considering the change of the human impact and of its potential effect on the soil, related to the real land use.

Within the class *Abandonment* we included both the change of any agricultural area versus natural or semi natural ones (agricultural abandonment) and urban or artificial structures (reservoirs, lakes). Class *Extensification* includes every changed area from an agricultural use with high human impact to another one with a lower impact. Detection of areas with the same land-use class during a transition period, were classified as *No change*. With the term *Persistence* we defined the change class of areas with transition from an agricultural use to another one without notable implication regarding the increment of human impact (for instance from *arable lands* to *olive groves*). *Intensification* includes every changed area from a natural or semi natural area in an agricultural area and the transition from an agricultural use toward another one with higher human impact but without remarkable effects on soil alteration. Within the class *Anthropization* we included any agricultural or natural surface, changed in areas where the human impact is potentially so high as to cause considerable alteration of soils. Within the class *Urbanization* we detected any land-use change involving the transformation of an area into an artificial surface. This class identifies, in the time, the soil consumption that is the severest form of soil degradation.

The analysis of the land-use maps (Fig. 7.2) and of the corresponding values in ha (Table 7.3) allow us to identify the following phenomena:

- in 1966 the arable lands represented almost 50 percent of the whole landscape of Mazzarrone but, from 1997, they completely disappear;
- the lack of espalier or sapling system vineyards and reservoirs and artificial lakes in 1966;
- the huge development of transformed lands and trellis system vineyards that from 45.1 ha (in 1996) grown up to 1957.8 ha in 2008

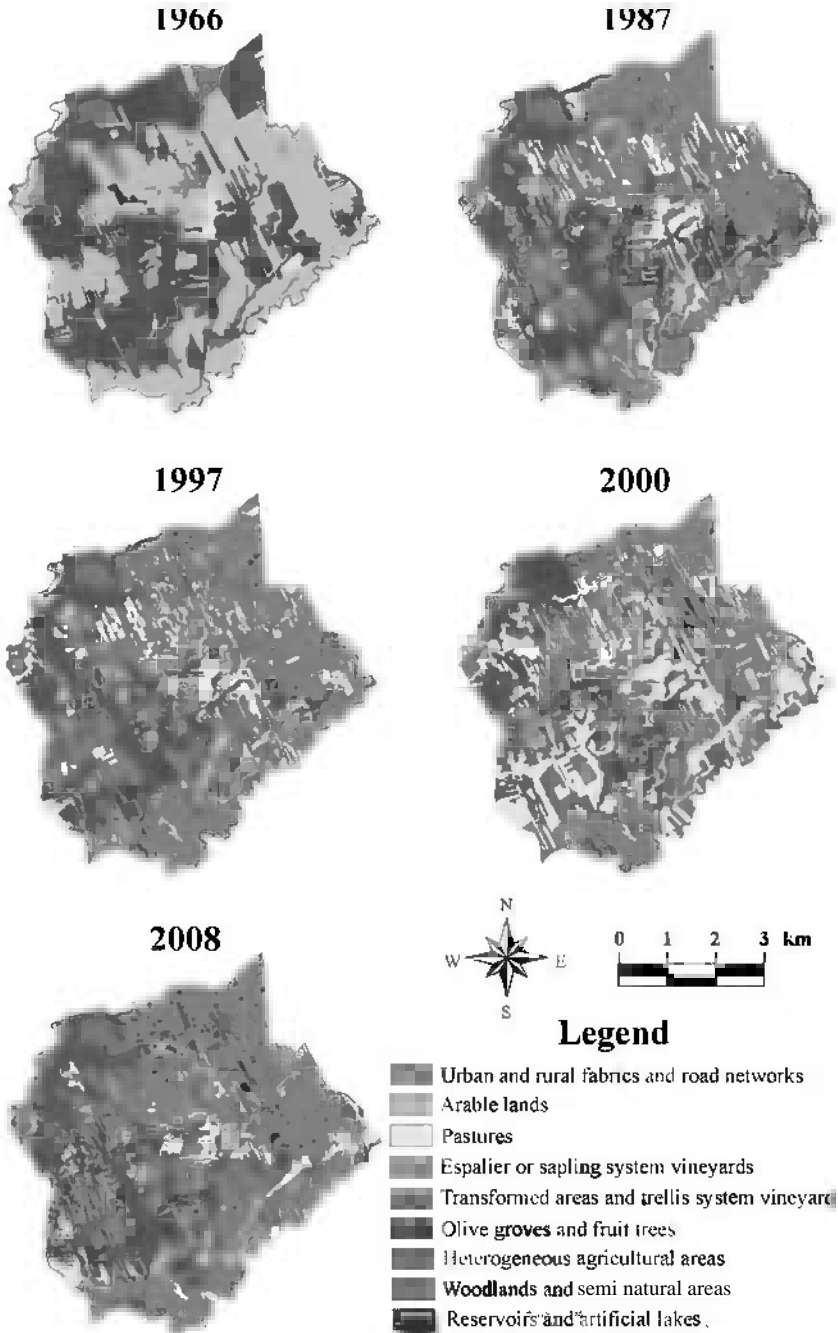


Figure 7.2 Land-use maps of the study area in 1966, 1987, 1997, 2000 and 2008.

Table 7.3 Land use in each investigated year.

Land use class	1966	1987	1997	2000	2008
	ha	ha	ha	ha	ha
Urban and rural fabrics and road networks	55.3	100.9	115.2	117.8	119.8
Arable lands	1,638.6	33.4	0.0	0.0	0.0
Pastures	70.5	441.5	429.3	884.5	325.6
Espalier or sapling system vineyards	0.0	29.4	34.9	25.1	60.0
Transformed areas and trellis system vineyards	45.1	1 704.1	1,906.6	1,631.2	1,957.8
Olive groves and fruit trees	851.8	598.2	412.2	454.0	446.4
Heterogeneous agricultural areas	40.4	40.2	40.0	0.0	48.4
Woodlands and semi natural areas	755.3	496.0	467.5	279.4	420.9
Reservoirs and artificial lakes	0.0	13.3	51.3	65.0	78.1
Total area	3,457.0				

(actually more than 50 percent of the whole landscape of Mazzarrone, i.e., almost the same surface of the arable lands of 1966);

- the continuous increment of the urbanized areas and of the reservoirs and artificial lakes;
- the variable increment in time of pastures, starting from few ha in 1966 and reaching the highest value in 2000 (884.5 ha) occurring simultaneously with the decline in heterogeneous agricultural areas (which were present in the map of the previous years);
- woodlands and semi natural areas show a decreasing trend in time with the lowest value in 2000 that increases again in 2008.

In 1966, the landscape of Mazzarrone was marked by the presence of *arable lands, olives groves and fruit trees* and *woodlands and semi natural areas*. The spatial pattern distribution of these predominant classes, simplified the landscape structure (a low fragmentation and number of patches). From 1987 the landscape complexity already became quite different and increased markedly.

The interpretation of the land-use change maps (transition maps) (Fig. 7.3) and of the relative transition matrixes for any period (Fig. 7.4) allow for the weighted reclassification of the Mazzarrone landscape and for a qualitative and quantitative spatial evaluation of the modification or persistence of the land use.

The transition maps show that most notable land-use changes took place from 1966 to 1987. In this period the unchanged area is relatively low ($\cong 30\%$ of the whole area) consisting mainly of *olive groves and fruit trees* and *woodlands and semi natural areas*. Even less is the percentage of the *Persistence* class ($\cong 7\%$), due mainly to the transformation of *arable lands* in *olive groves and fruit trees*. The *Extensification* class reaches in this period a value of $\cong 10\%$, due to the conversion of both 206.4 ha of *arable lands* and 111.3 ha

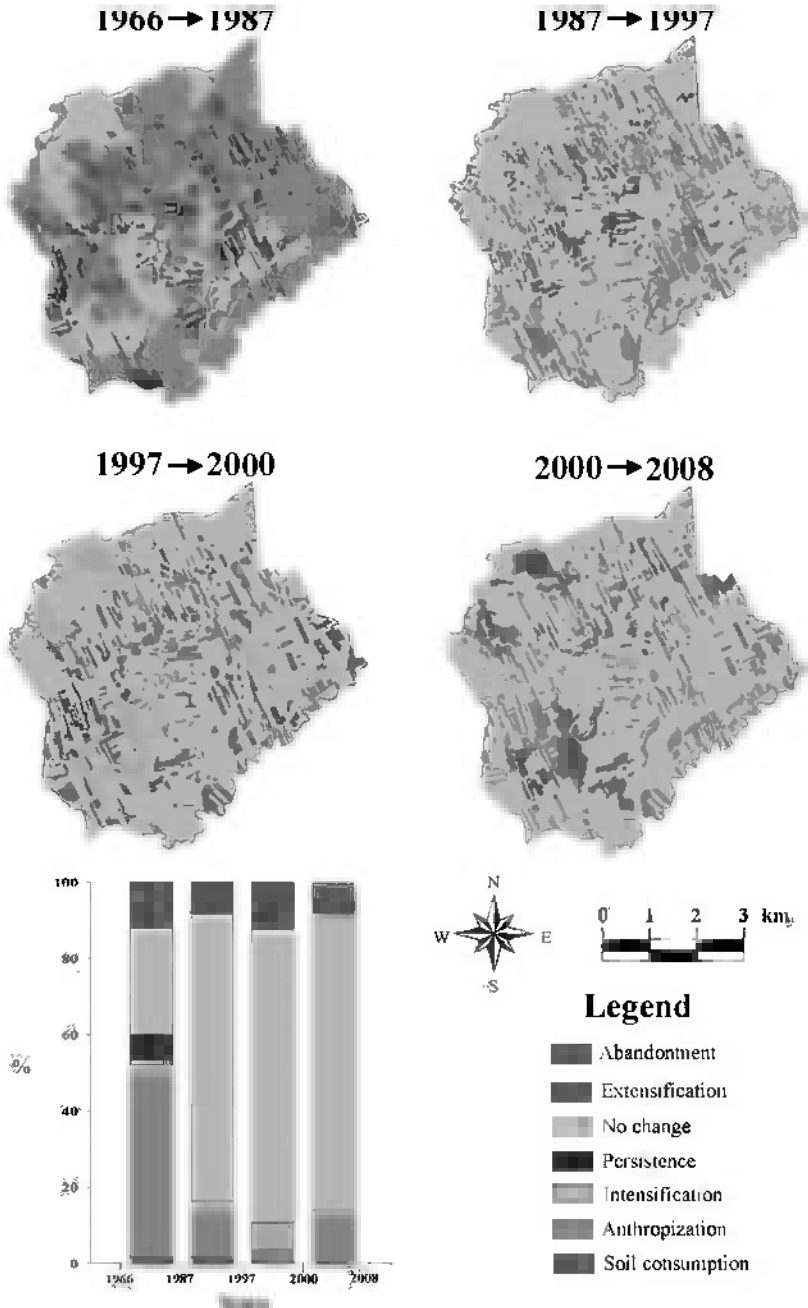


Figure 7.3 Land-use change maps from 1966 to 2008 reclassified according to the established transition model and proportion of each class for transitional periods.

	Urban and rural fabrics and road networks	Arable lands	Transformed areas and trellis system vineyards	Espalier or sapling system vineyards	Olive groves and fruit trees	Heterogeneous agricultural areas	Pastures	Woodlands and semi natural areas	Reservoirs and artificial lakes
1966 → 1987	52.8	0.0	0.9	0.0	1.4	0.0	0.1	0.0	0.0
Urban and rural fabrics and road networks	21.5	33.0	1106.1	14.3	259.5	0.0	206.4	20.7	7.2
Arable lands	0.3	0.0	42.9	0.0	0.1	0.0	1.4	0.0	0.4
Transformed areas and trellis system vineyards	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Espalier or sapling system vineyards	23.5	0.2	288.2	4.4	351.2	0.0	111.3	68.8	4.3
Olive groves and fruit trees	0.0	0.0	0.1	0.0	0.0	40.2	0.0	0.0	0.1
Heterogeneous agricultural areas	0.2	0.0	21.8	8.9	0.0	0.0	37.0	2.6	0.0
Pastures	2.3	0.3	244.1	1.9	16.2	0.0	85.5	410.8	1.3
Woodlands and semi natural areas	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Reservoirs and artificial lakes									
1987 → 1997	96.7	0.0	1.4	0.0	0.5	0.0	1.6	0.0	0.1
Urban and rural fabrics and road networks	0.0	0.0	30.4	0.0	0.0	0.0	0.0	1.8	1.1
Arable lands	11.2	0.0	1482.5	21.8	30.3	0.0	124.7	8.9	26.3
Transformed areas and trellis system vineyards	0.1	0.0	13.5	9.4	0.1	0.0	4.0	0.0	0.1
Espalier or sapling system vineyards	6.3	0.0	163.6	3.4	338.4	0.0	58.1	22.8	6.8
Olive groves and fruit trees	0.0	0.0	0.0	0.0	0.0	40.0	0.2	0.0	0.0
Heterogeneous agricultural areas	2.5	0.0	204.9	0.4	26.0	0.0	197.9	4.6	4.8
Pastures	0.3	0.0	8.7	0.0	14.4	0.0	41.8	436.4	1.3
Woodlands and semi natural areas	0.0	0.0	1.6	0.0	0.3	0.0	0.8	0.0	10.8
Reservoirs and artificial lakes									
1997 → 2000	109.2	0.0	2.6	0.0	0.3	0.0	2.7	0.0	0.4
Urban and rural fabrics and road networks	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Arable lands	3.2	0.0	1524.0	3.3	31.7	0.0	326.2	0.9	11.0
Transformed areas and trellis system vineyards	0.0	0.0	4.4	20.8	8.8	0.0	1.2	0.0	0.0
Espalier or sapling system vineyards	1.5	0.0	20.4	0.1	334.4	0.0	58.2	0.9	1.8
Olive groves and fruit trees	0.0	0.0	0.0	0.0	40.0	0.0	0.0	0.0	0.0
Heterogeneous agricultural areas	3.8	0.0	74.6	0.8	13.2	0.0	333.1	1.4	2.2
Pastures	0.0	0.0	2.8	0.0	26.0	0.0	162.6	283.1	0.0
Woodlands and semi natural areas	0.0	0.0	2.5	0.0	0.0	0.0	0.5	0.1	49.5
Reservoirs and artificial lakes									
2000 → 2008	116.2	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.2
Urban and rural fabrics and road networks	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Arable lands	1.3	0.0	1525.6	41.7	49.7	0.0	1.3	2.8	9.2
Transformed areas and trellis system vineyards	0.0	0.0	10.4	14.3	0.3	0.0	0.0	0.0	0.0
Espalier or sapling system vineyards	0.1	0.0	19.7	2.6	363.4	39.9	2.1	20.4	0.9
Olive groves and fruit trees	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heterogeneous agricultural areas	2.8	0.0	399.2	1.3	32.6	0.0	322.2	128.0	4.2
Pastures	0.0	0.0	0.8	0.0	0.2	8.4	0.0	276.7	0.2
Woodlands and semi natural areas	0.0	0.0	0.9	0.0	0.1	0.0	0.0	0.0	63.4
Reservoirs and artificial lakes									

Legend

- Abandonement
- Persistence
- Intensification
- Soil consumption
- Extensification
- No change
- Anthropization

Figure 7.4 Transition matrixes of the land-use classes, for every period, and relative thematic classification in transition classes. Values are in ha.

of olive groves and fruit trees to pastures. Intensification class covers an area of $\approx 3.5\%$ due to the conversion of woodlands and semi natural areas to pasture or olive groves and fruit trees. The Anthropization class covers more than 50 percent of the Mazzarrone area: 1106.1 ha of arable lands plus 288.2 ha of olive groves and fruit trees and 244.1 of woodlands and semi natural areas, were

transformed in *trellis system vineyards*. Also *soil consumption* in those years was notable: 61.23 ha to favor the urban development and supplying water was always to the detriment of *arable lands, olive groves and fruit trees and woodlands and semi natural areas* (28.7; 27.8 and 3.6 ha respectively).

Between 1987 and 1997 there is an almost uniform distribution of the transition classes, except the *No change* class (2612.1 ha) involving mainly the *trellis system vineyards* and secondarily *woodlands and semi natural areas, olive groves and fruit trees* and *pastures*. Also in this period there is a notable *soil consumption* (60.8 ha) but in this case most of these areas were converted to the classes *transformed lands and trellis system vineyards*. It is significant to note that from this transition period onwards, the *Persistence* class was strongly reduced due not only to the disappearance of the *arable lands* but also to the unfavorable economics to transform agricultural areas, with equivalent profits, with the possibility to transform these into *trellis system vineyards* (to get higher profits). Also *Anthropization* shows a decrement (lowest value in 1997/2000) altogether varying around an average value of $\cong 10\%$. Both *Extensification* and *Intensification* show their higher values in 1997/2000 transition period, due respectively to the conversion of *trellis system vineyards* to *pastures* (362.2 ha) and to the conversion of *woodlands and semi natural areas* to *pasture*. However, most of the same converted areas turn again in the previous temporal land-use class in the next transition period (2000/2008). These returns back in the land use are detected as *Anthropization* and *Abandonment* respectively. In the first case the change is probably due to the temporary abandonment of the vineyards between the explant and the implant of new vines; in the second there may have been forestry utilization or a fire followed by natural or anthropic restoration of the vegetation. Considering the entire analyzed period (from 1966 to 2008) the natural and semi natural areas decreased by 334.4 ha. The change resulted in, for most of those areas (244.1 ha), the conversion to *transformed areas and trellis system vineyards* (*Anthropization*) during the 1966/1987 transition period and the remaining part to agricultural intensification.

This result is significant and gives evidence to the opposite direction of the actual Italian national trend (i.e., an increase of the semi natural and natural areas favored by an environmental protection policy).

Excluding the 2000/2008 transition matrix, *Abandonment*, in the whole period, regarded mainly the conversion of *olive groves and fruit trees* to *woodlands and semi natural areas*. *Urban* or reservoir and lakes were dismissed as being irrelevant. The first one was due to the dismantling of temporary artificial structures (barns or small old shelters), the second one mainly to the removal of small out-surface reservoirs.

Overlaying all the land-use maps and querying the GIS database, it was possible to point out the main "trajectories of change" (Mertens and

Lambin 2000), defined as temporal sequences of the nine successive land-use types at each patch at five time points from 1966 to 2008.

The most significant trajectories, both in terms of number of patches (frequency) and total area (magnitude), were:

II-V-V-V-V

VI-V-V-V-V

VIII-V-V-V-V

that involve an area of 718.0, 176.3 and 182.4 ha with 260, 122, and 29 patches respectively.

This means that most of *arable lands* (II), *olive groves* (VI) and *woodlands and semi natural areas* (VIII) in 1966/1987 transition period were changed into *trellis system vineyards* (V) and so maintained till to the recent time point (2008).

Other very frequent trajectories were:

II-V-V-III-V

II-V-III-V-V

II-V-III-III-V

Such trajectories characterize those areas changed into *trellis system vineyards* in 1966/1987, then for some years were temporarily abandoned (we defined them as *pastures* (III)) and changed back into vineyards.

More than 120 ha in total follow the trajectories:

VI-VI-V-V-V

II-VI-V-V-V

meaning that *olive groves and fruit trees* from 1987 were widely converted into *trellis system vineyards* till to 2008, and after that the complete disappearance of *arable lands*.

The most consistent transformations concerned the conversion over time of some land-use classes towards *transformed areas and trellis system vineyards* (*Anthropization*). This particular land use, as mentioned before and the soil survey has demonstrated, involves modifications in the soils so pronounced as to completely change their nature and features. Considering the disturbing effects produced by the *Anthropization* on the original soils, this is considered as an irreversible process in space. The correspondence between the presence of *trellis system vineyards*, or temporary *transformed areas*, and "human made soils" (i.e., Anthrosols) is direct and one-to-one. Actually, for the local agricultural marketing and the high transformation

costs for the anthropogenic soil formation, only this particular crop system, which allows a high-quality production, can be economically advantageous and attains the highest profits.

Detection by remote sensing of *transformed lands or trellis system vineyards* points out the presence of anthropogenic soil processes. Thus, extrapolating from every land-use map those areas classified as *transformed lands or trellis system vineyards*, we assessed, by overlaying in the GIS, the potential spatial diffusion of the anthropogenic areas for the 1966/2008 period. The produced map (Fig. 7.5) shows that in time, 65 percent of the total land was exposed to soil anthropization process. The map also records the distribution patterns of soil consumed in the last point in time (2008) for both *Urbanization* and *reservoirs and artificial lakes* establishment (131 and 82.6 ha respectively). The total percentage of this area is not negligible with regard to the whole study area (6.1 percent); more significant is the rapid increase from 1966 of the urban area (more than twice) due mainly to the intensification of residential, rural and commercial fabrics. Considering the remarkable improvement in the economic conditions of the local population due to the high profits in *trellis system vineyards* (Lo Verde 1995), the increasing urbanization seems to

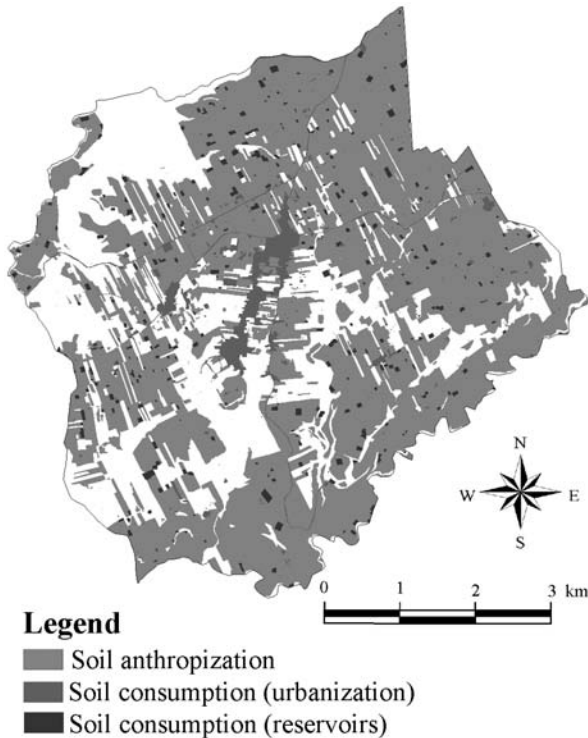


Figure 7.5 Patterns of soil anthropization and soil consumption in Mazzarrone till 2008.

be linked directly to the anthropization process. *Reservoirs and artificial lakes* once not present, increase proportionally to the anthropic areas in time and is directly linked to the anthropization, because *trellis system vineyards* are the sole land use requiring irrigation: their spatial pattern also follows in time the spatial distribution of the *trellis system vineyards*. In other words, the anthropization process has not only led to the anthropogenic soil formation but has also affected the soil consumption rate.

A thorough comprehension of temporal land-use change by analysis of the transition matrices (quantification of change) and trajectories (directions and trends of change) was useful to identify some driving forces related to the anthropogenic soils pattern.

Transition matrices show that the landscape of Mazzarrone was already substantially changed in 1987. In this period most of land was exposed to anthropization. The first widespread conversions concerned mainly *arable lands* towards *trellis system vineyards* and *olives groves and fruit trees* were also converted consistently after the disappearance of *arable lands*. This process probably follows some economic logic. In fact, *arable lands* in semi arid environments, which receive the lowest profits, were all promptly converted into profitable vineyards. *Olive groves and fruit trees*, unlike *arable lands*, generate higher profits but the conversion costs are also higher due to the additional charges for the trees explants, and they were converted more slowly.

A consistent rate of *woodlands and semi natural areas* were converted into *trellis system vineyards* exclusively in 1966/1987. Spatial overlay of these transformed areas and DEM shows that the transformations occurred mainly on the more favorable land (flat or gentle slope). Also querying the GIS database for the land cover units, the transformed areas were mainly garrigues or sparsely vegetated areas. Woodlands or shrub-lands were not converted on steeper slopes, probably because of higher transformation costs (trees or tree-like explant). In the next transition period the remaining area was not further affected by anthropization but did experience a conversion towards *olive groves and fruit trees* or *pastures*.

Pastures, such as *olive groves and fruit trees*, were consistently transformed in anthropic areas during 1987/1997. *Pastures* detected after 1997 result from the conversion of *trellis system vineyards*. However, as indicated in the change trajectories and confirmed by field surveys, these *pastures* were transformed into vineyards requiring normally a 20 yr production cycle. The time of this temporary abandonment can be even more than 5 yr. This could represent a gap in the detection of *trellis system vineyards* by remote sensing, especially when the observation frequency is low. For example, if a *trellis system vineyard* is detected during the abandonment repeatedly over time then it will be recognized in the land-use map as pasture, even

though it was really affected by anthropization. In consequence, this gap could falsify the correct size of anthropogenic soils by underestimation.

Analysis of transition matrices and change trajectories proved useful to describe quantitatively land-use change patterns and have been used for several studies on modeling dynamics of land-use change (Aaviksoo 1993, Li 1995, Cousins 2001). Land use transition matrices and trajectories have been used for making projections of diffusion of anthropogenic soils in the future and provide a better understanding of the possible effect on pedodiversity.

4. Effects of Land use on Anthropogenic Soil Formation (pedotechniques)

To investigate the genesis of the anthropogenic soils formation due to the land-use change over time on the landscapes of Mazzarrone, we focused our attention on two neighboring fields in North-East Mazzarrone (Fig. 7.6). The first (500 x 40 meters) had natural soils because it had never been cultivated; the second (500 x 80 meters) showed anthropogenic soils

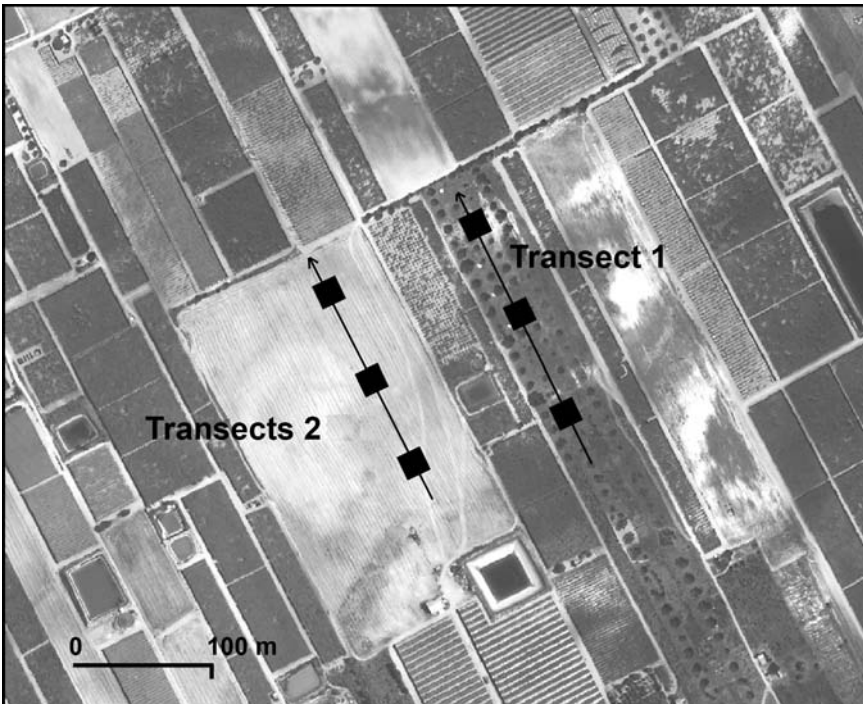


Figure 7.6 Location of the transects surveyed in the Mazzarrone study area. The black squares indicate the position of the soil profiles.

but before the conversion to vineyards showed the same natural soils as the uncultivated neighboring field. In the first field, chosen as test area, in June 2003 a transect with three pedons (*Transect 1*) was surveyed and sampled according to the sequence of the pedogenetic horizons (USDA-NRCS 2002). A similar transect in the second field (*Transect 2*) was surveyed and sampled in July 2004, according to the ICOMANTH indications (2004).

The natural soils of the test area evolve on marly limestone substrata, show an average deep A-C or A-Bw-C profile, with a basically loamy texture (clay from 165 to 385 g kg⁻¹), a good sub-angular blocky structure and a sub-alkaline reaction (pH 7.7 ÷ 7.9). In the soil solum, carbonates are moderately present, both total (from 16 to 510 g kg⁻¹) and active (from 7 to 169 g kg⁻¹), and the organic carbon decreases regularly with the depth. Following the Soil Taxonomy rules (Soil Survey Staff 2010), these pedons (that can be considered as the soils originally present in both the fields) can be classified as *Entic Haploxerolls*. These pedons that evolve in a xeric environment, show an epipedon that fits all the requirements of a mollic. Moreover they are characterized by little development in the subsoil, lie on late-Pleistocene deposits and have free carbonates throughout the cambic horizon or in all parts of the mollic epipedon below a depth of 25 cm from the mineral soil surface.

In July 2004, three soil profiles were described and sampled to evaluate the impacts of agricultural management over the last several decades on the wider field. The wider field that in origin was characterized by the same vegetation, by the same marly limestone substratum and by the same soils of the test area (as testified by previous soil survey and by the aerial photo interpretation), in 1984 was subjected to a first incisive plowing for planting a trellis system vineyards. Trenching was 90–100 cm deep carried out with a mold-boarded one-furrow plow, which gave complete overturning and deep stirring up of the horizons. Such agricultural management transformed the original soils (*Entic Haploxerolls*) in Arents, more specifically to *Haplic Xerarents*.

In spring 2003 the morphology of this field was gently reshaped by covering the surface of the anthropic Entisols with a layer of marly limestone ranging from 50 to 70 cm in thickness that (ICOMANTH 2003) can be considered as a human transported material (HTM). The surface molding aimed to make the slope more gentle and to increase the amount of carbonates and of the value and chroma of the soil color. Actually, a greater amount of carbonates has a positive effect on the quality of the fruit because it increases the “crackling” of the grape and influences the albedo of the soil with positive effects on the amount of sugar in the fruits. In August 2003, the Entisols covered with the HTM layer, were deep trenched with a mold-board one-furrow plow, to around 100 cm in depth and, in July 2004

a transect of three soil profiles were described and sampled following the ICOMANTH advices (2004).

From a morphological point of view, the most striking feature of the soils of the transect was represented by a double sequence of horizons at an oblique angle to the soil surface. The soil color, particularly in the topsoil, became very light (from light gray, 10YR 7/2 to white, 2.5Y 8/2). The amount of carbonates, both total (from 415 to 759 g kg⁻¹) and active (from 165 to 187 g kg⁻¹) became very high. The organic carbon amount remained low and decreased irregularly with the depth (from 1 to 7 g kg⁻¹).

To classify these highly disturbed soils (Fig. 7.7) following the Soil Taxonomy rules, we must consider that “significant changes in the nature of the soil by humans cannot be ignored” (Soil Survey Staff 1999). This is particularly true in our case where the soils prepared for vineyard cultivation, were so deeply modified not only by covering the pre-existing Haplic Xerarents (obtained from pre-existing Entic Haploxerolls) with a layer (50–70 cm) of marly limestone but also by mixing the soils to 100 cm with the aid of heavy machinery. This way of building soil excludes the possibility of classifying such soils as Arents. As a consequence and following the Soil Taxonomy criteria, the anthropogenic soils we surveyed, should be classified in a new soil Order which can be reasonably called *Anthrosols*. If we consider “ant” as the “formative element” in the name of this new soil Order (ant, from Greek *anthropos* = man) and bearing in mind that the pedo-climatic features of our study area are of Mediterranean type, at a suborder level we can classify such anthropic soils as *Xerants*. Moreover, considering that these soils were made up of a mixture of inorganic materials (mainly marly limestone) transported and deposited through earth excavations, covering a pre-existing very deeply disturbed soil (the Haplic Xerarent after the first intervention of man), *Geofragmexerant* could be the most suitable name at Great Group level (from Greek *geomai* = to become hearth, and from Latin *fragmenta* = spoils). Considering that the surface was mixed to a depth of about 100 cm with the aid of heavy machinery before the second vineyard plantation, at Sub-Group level they can be regarded as *Miscic Geofragmexerant* (from Latin *miscere* = to mix) meaning that such anthropogenic soils are characterized by disturbed materials in the whole profile.

5. Spatio-temporal Distribution of Anthropogenic Soils

To further consider the human factor in modifying and altering the soilscape of the Mazzarrone area, we used the findings of a sociological and ethnological assay (Lo Verde 1995) together with temporal data from unpublished soil surveys carried out from 1964 to 2008. Such soil surveys were supported by aerial photo interpretation and land-use maps in the



Figure 7.7 Anthropogenic soil profile in Mazzarrone.

years 1955, 1966, 1987, 1997, 2000, 2008, providing six temporal soil maps (Fig. 7.9). We validated these maps in field through the examination of several soil profiles. The map legend of each soil map reports the spatial

distribution of the soils classified at subgroup level according to the Soil Taxonomy.

Geographic Information System (GIS) technology was used to manage and analyze changes in land use and soil patterns over time. All datasets were archived and managed in raster format with 1-meter spatial resolution. Multi-temporal analysis was used to assess the soilscape change which was affected by the deep soil transformation due to large scale farming. A stochastic simulation, which coupled Markov chains and Cellular Automata (CA), was set up to model the spatial-temporal change of the soil pattern and to predict the future evolution of the soilscape.

Markov chains (called Markov modeling, or Markov analysis, for brevity) is an aggregate, stochastic, modeling process. Such a simulation technique has been applied since the mid 1960s in the analysis of land-use change, mainly in the study of land conversion processes mostly in urban contexts such as suburbanization, neighborhood housing turnover (Clark 1965, Drewett 1969, Gilbert 1972, Bell 1974, 1975, Bell and Hinojosa 1977), and in analyzing the historical dynamics of urbanization in agricultural areas (Muller and Middleton 1994). More recently, Markov analysis has been applied to problems of assessing the impacts of projecting changes in organic carbon stores caused by land-use changes (Howard et al. 1995, Prasad et al. 2003, Schneider 2007).

Conceptually, Markov analysis calculates, for a system of categorical states (in our case soil classes, i.e., S_1, S_2, \dots, S_n , corresponding to soil map unit/soil subgroups spatially constituted in the GIS by cells), the transition of one state to another with some probability which depends only on the current state. The probability P to move cells from a class i to another class j is called a transition probability. Markov analysis uses these probabilities in the form of a transition matrix:

$$P = \begin{bmatrix} P_{11}, P_{12}, P_{13}, \dots, P_{1n} \\ P_{21}, P_{22}, P_{23}, \dots, P_{2n} \\ P_{31}, P_{32}, P_{33}, \dots, P_{3n} \\ \dots \\ \dots \\ P_{n1}, P_{n2}, P_{n3}, \dots, P_{nn} \end{bmatrix}$$

We computed the transition probabilities on the basis of the soilscape change over a 42 yr period, from 1966 (in which we observed for the first time the presence of Anthropogenic soils) to 2008, and we predicted the potential soilscape change in 2050, i.e., the next 42 yr from 2008 using the same probability observed during man’s intervention by large scale farming

on soils (1966–2008). In short, according to Logsdon et al. (1996) we created probability maps within GIS to model the past soilscape change in space and its projection into the future.

For our specific purposes, we set the transition probability for each soil class as “null” to transition to in another soil class, except for the Anthropogenic soil class which can freely occupy the space of any other soil class according to the relative transition probability map. We also excluded from our simulation model the surface occupied by reservoir constructions and urban areas.

As the Markov analysis is space insensitive (i.e., having no sense of the geographical feature space) and provides only accurately the transition probability for each particular soil class as a whole, we coupled the Markov analysis with a two dimensional GIS-based Cellular Automata algorithm which returns a spatial dimension to our model.

Generally, Cellular Automata (CA) consists of a spatial filter in which each cell can assume a discrete state at any one time. All cells, each one in a particular position in a time $t=0$, change state simultaneously as a function of their own state, of the state of the cells in their neighborhood and in accordance with a specified set of transition rules (Engelen et al. 1995, White and Engelen 1997, Batty et al. 1999). In our model, we use a set of quantitative transition rules in Cellular Automata (CA) which in sequence of priority are: i) the transition probabilities maps originated by the Markov analysis and ii) a specific map of weights to address the spatial transition of the anthropogenic soils into other soil classes in the predicted time.

To fulfill this map of weight, we used a set of sub-rules based on the analysis of: i) the land-use change transitions in the years 1955-1966-1987-1997-2000-2008, ii) the land-use pattern in 2008, iii) the most recent orthophotos, iv) the anthropogenic soil pattern in soilscape map 2008, and v) the slope map derived from a DEM with 20 meters of spatial resolution.

From the land-use map of 2008, we selected only those types of land-use classes which the maps of change in the observed past periods indicated transformation, at least once in the vineyards. We assume those areas to be suitable for further change into vineyards and consequently to be areas in which soils are potentially altered by anthropogenic processes due to large scale farming. We excluded also *a priori* all the forested areas because they are regulated by law, which preserve them from any change of land use. In a further step we segmented into parcels those areas following the physical pattern observed on the recent orthophotos, showing the land fragmented by agricultural/property unit. In a successive step we analyzed the pattern of the anthropogenic soils in 2008 over the slope map, we extracted an optimality distribution curve. Finally, we assigned to every parcel in our selected areas a specific weight (ranging from 0 to 1) calculated by normalizing the mean slope value of the parcels according

to the optimality curve (Fig. 7.8). We wish to highlight that our model is only physically based and is independent of economic drivers which can surely affect land-use change and consequently lead to soil transformation. The model ignores socioeconomic forces and processes, which produced the observed patterns and could change in the future, and it takes into account only the correlations observed between the land use/soil patterns and some physical factors. In CA sub-model we set the window of the neighborhood contiguity filter with a value of 10 cells which correspond to 10 meters on the ground.

We analyzed of the transformation of the original soils to anthropogenic ones, as well as the loss of agricultural areas and soil sealing due to urban expansion and construction of reservoirs. Table 7.4 illustrates that 62.5 percent of the total soil coverage was transformed between 1955 and 2008. The proportion of transformation consisted of 57.8 percent by large scale farming, 2 percent by urban sealing (housing and road construction) and 2.7 percent by water reservoir sealing. Proportions of the soil subgroups show that the most transformed soils belonged to the order of Mollisols followed by Inceptisols and Alfisols. Entisols were less transformed, while Vertisols were not affected by any transformation.

The Markovian transition probabilities matrix for the next 42 yr, calculated on the basis of the soilscape changes in the period 1966–2008, shows the probability value that every soil class in our geographical space has to transition to a different class. The Markov analysis assumes that the probability values range from 0 to 1 for each class (i.e., soil types). Markov transition matrix shows the highest probability for three specific kind of soils which are, in decreasing order of probability, Calcic Haploxerolls, Typic

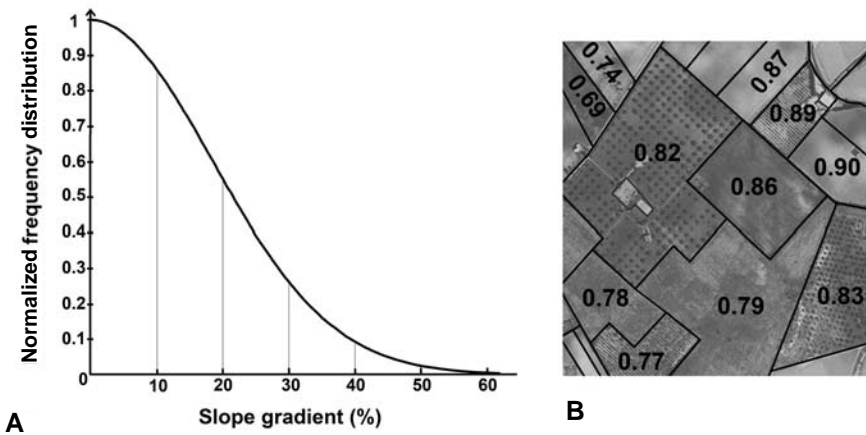


Figure 7.8 A) Optimality curve of the Anthropogenic soils in 2008 by slope gradient. B) Example of the map of weights overlaying the most recent orthophotos.

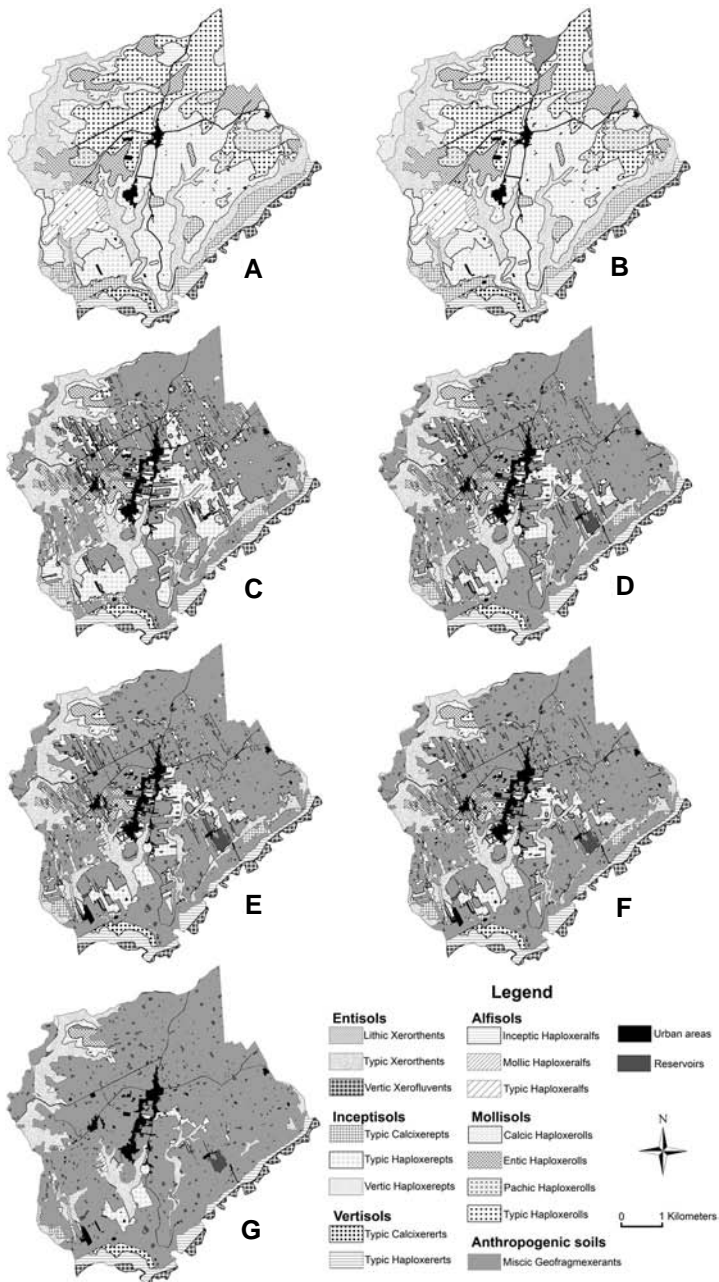


Figure 7.9 Soil maps of the Mazzarrone area in 1955 (A); 1966 (B); 1987 (C); 1997 (D); 2000 (E); 2008 (F) and predicted for 2050 (G). Anthropogenic soils are classified according to the proposal by Dazzi and Monteleone (2007).

Table 7.4 Transformation (in hectares and percentage) of pre-existing soils into anthropogenic soils by large scale farming and soil sealing by urban and water reservoir construction until to 2008.

Original soils ¹	Miscic Geofragmexerants ²	Urban Sealing ³			Reservoir Sealing ⁴	Total			
		ha	%	ha		%	ha	%	ha
Entisols									
Lithic Xerorthents	3.0	21.6	0.0	0.0	0.0	0.0	3.0	21.6	
Typic Xerorthents	118.0	21.8	4.0	0.7	5.0	0.9	127.0	23.4	
Vertic Xerofluvents	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
Inceptisols									
Typic Calcixerpts	145.0	75.5	4.0	2.1	2.0	1.0	151.0	78.6	
Typic Haploxerepts	565.0	57.7	43.0	4.4	36.0	3.7	644.0	65.7	
Vertic Haploxerepts	76.0	77.6	0.0	0.0	4.0	4.1	80.0	81.6	
Vertisols									
Typic Calcixererts	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
Typic Haploxererts	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
Alfisols									
Inceptic Haploxeralfs	99.0	82.5	0.0	0.0	3.0	2.5	102.0	85.0	
Mollic Haploxeralfs	2.0	18.2	0.0	0.0	0.0	0.0	2.0	18.2	
Typic Haploxeralfs	106.0	73.1	1.0	0.7	7.0	4.8	114.0	78.6	
Mollisols									
Calcic Haploxerolls	37.0	88.1	1.0	2.4	1.0	2.4	39.0	92.9	
Entic Haploxerolls	150.0	56.0	9.0	3.4	7.0	2.6	166.0	61.9	
Pachic Haploxerolls	109.0	84.5	1.0	0.8	7.0	5.4	117.0	90.7	
Typic Haploxerolls	557.0	87.7	7.0	1.1	19.0	3.0	583.0	91.8	
Total	1967.0	57.8	70.0	2.0	91.0	2.7	2128.0	62.5	

¹Before large scale farming.

²Anthropogenic soils originated by large scale farming classified according to the proposal by Dazzi and Monteleone (2007).

³Soil sealing by housing and road construction.

⁴Soil sealing by water reservoir construction.

Haploxerolls and Pachic Haploxerolls, to transit to the class of anthropogenic soils with a values ranging from 0.84 to 0.89. A second group of soils, with a high probability for transformation into antropogenic soils are Inceptic Haploxeralfs, Vertic Haploxerepts and Typic Haploxeralfs, with values ranging from 0.78 to 0.73. Typic Haploxerepts and Entic Haploxerolls also have high conversion probabilities, with values of 0.57 and 0.55, respectively. It should be noted that Vertic Xerofluvents, Typic Haploxererts and Typic Calcixererts have nil probability to be transformed to another class; in the past they were not affected by anthropogenic processes or by sealing.

The Cellular Automata (CA) submodel simulation regulated by specific rules addresses the prediction on the basis of the Markovian transition probabilities into the geographical space, defining a potential soilscape map of prediction in the future 2050 (Fig. 7.9G).

A comparison of the predicted map in 2050 with the field map in 2008, shows that probably, the soilscape of Mazzarrone will lose the Calcic Haploxerolls, the Typic Haploxerolls and the Pachic Haploxerolls, which could be entirely transformed in anthropogenic soils (Table 7.5). It would also be of interest as to the extent of Typic Calcixerepts, Vertic Haploxerepts, Typic Haploxeralfs and Inceptic Haploxeralfs (Table 7.6) which are reduced to a few hectares and probably, in a little more long term future scenario, be completely transformed.

Table 7.5 Simulated transformation (in hectares and percentage) of original soils, present in 2008, into Anthropogenic soils by 2050.

Original soils ¹	Miscic Geofragmexerants ²	%
Entisols		
Lithic Xerorthents	0	0.0
Typic Xerorthents	46	11.1
Vertic Xerofluvents	0	0.0
Inceptisols		
Typic Calcixerepts	33	80.4
Typic Haploxerepts	210	62.5
Vertic Haploxerepts	11	61.1
Vertisols		
Typic Calcixererts	0	0.0
Typic Haploxererts	0	0.0
Alfisols		
Inceptic Haploxeralfs	17	89.4
Mollic Haploxeralfs	0	0.0
Typic Haploxeralfs	22	80
Mollisols		
Calcic Haploxerolls	3	100
Entic Haploxerolls	78	76.5
Pachic Haploxerolls	12	100
Typic Haploxerolls	52	100
Total	484	14.9

¹Preserved from Anthropogenic process until 2008.

²Anthropogenic soils originated by large scale farming classified according to the proposal by Dazzi and Monteleone (2007).

Table 7.6 Changes of soil map classes in the study area from 1955 to 2050.

Orders	Subgroup	1955	1966	1987	1997	2000	2008	2050
Entisols		ha						
	Lithic Xerorthents	14	14	12	11	11	11	11
	Typic Xerorthents	542	542	460	425	423	414	368
	Vertic Xerofluvents	71	71	71	71	71	71	71
Inceptisols								
	Typic Calcixerepts	192	191	84	48	46	41	8
	Typic Haploxerepts	980	979	575	393	362	336	126
	Vertic Haploxerepts	98	94	39	29	24	18	7
Vertisols								
	Typic Calcixererts	27	27	27	27	27	27	27
	Typic Haploxererts	132	132	132	132	132	132	132
Alfisols								
	Typic Haploxeralfs	145	145	50	36	35	31	9
	Inceptic Haploxeralfs	120	99	41	22	19	19	2
	Mollic Haploxeralfs	11	11	10	9	9	8	8
Mollisols								
	Typic Haploxerolls	635	615	146	76	63	52	0
	Calcic Haploxerolls	42	41	6	3	3	3	0
	Entic Haploxerolls	268	267	139	109	106	102	24
	Pachic Haploxerolls	129	129	32	16	12	12	0
Anthropogenic soils								
	Miscic Geofragmexerants ¹	0	45	1520	1875	1916	1967	2451
	Urban areas ²	51	54	100	109	119	121	121
	Reservoirs ²	0	0	13	66	79	92	92
	Total	3,457	3,457	3,457	3,457	3,457	3,457	3,457

¹Classification proposed for Soil Taxonomy system according to Dazzi and Monteleone (2007).

²The surface of the urban area and the reservoirs does not change in predicted 2050 because it was excluded in the space-time simulation.

6. Human Influence on Soilscape Diversity

The pedodiversity in the study area was assessed in different years at the soilscape level using soil maps of Mazzarrone, showing the original soilscape before any human intervention (1955), the soilscapes in the years 1966, 1987, 1997, 2000, 2008 and the predicted soilscape by space-time simulation in 2050. We consider soil types at suborder taxonomy level as classes constituting the Mazzarrone soilscape.

Several indices have been used by the scientific community to assess soil diversity, most of which have been borrowed from landscape ecology (Richter and Babbar 1991, Ibáñez et al. 1995, 1998, 2005a,b, Guo et al. 2003a,b, Saldaña and Ibáñez 2004, 2007, Toomanian et al. 2006, Phillips and Marion 2007, Krasilnikov et al. 2009, Costantini and L'Abate 2009). These are traditionally used to measure spatially the landscape complexity and diversity. Many others have been developed in recent years specifically to measure the pedodiversity taking into account both the relative abundance and the taxonomic differences of the classes constituting a soilscape (McBratney and Minasny 2007, Minasny et al. 2010, Petersen et al. 2010).

The diversity of the Mazzarrone soilscape pattern over time was assessed exclusively using the Richness index(s), the Shannon's diversity index (SHDI), the Simpson diversity index (SIDI), the modified Simpson's diversity index (MSHDI), the Shannon's evenness index (SHEI), the Simpson's evenness index (SIEI) and the modified Simpson's evenness index (MSIEI). S, SHDI and SHEI are calculated in Chapter 1.

The Simpson Diversity Index is calculated as follows:

$$SIDI = 1 - \sum_{i=1}^n P_i^2$$

where P is the portion of soilscape occupied by a soil class i, and n is the total number of soil classes forming the soilscape. It is less sensitive to the presence of rare classes and has an interpretation that is much more intuitive than SHDI. It ranges from 0 to 1 (0 when the soilscape contains only one class, i.e., no diversity).

The Modified Simpson's Diversity Index is calculated as follows:

$$MSIDI = -\ln \sum_{i=1}^n P_i^2$$

where P is the portion of soilscape occupied by a soil class i, and n is the total number of soil classes forming the soilscape. It is less intuitive to interpret than SIDI; it belongs to a mathematical class of diversity indices

similar to SHDI. The MSIDI ranges > 0 with no limits and equals 0 when the soilscape contains only one class (i.e., no diversity).

The Simpson's Evenness Index is calculated as follows:

$$SIEI = \frac{1 - \sum_{i=1}^n P_i^2}{1 - \left(\frac{1}{n}\right)}$$

where P is the portion of soilscape occupied by a soil class i, and n is the total number of soil classes forming the soilscape. It expresses conceptually the complement of dominance. An even distribution of area among class types results in maximum evenness. SIEI ranges between 0 and 1 and, like SHEI, it is 0 when the soilscape contains only one class (i.e., no diversity) and assumes value 1 when the distribution of area among classes is perfectly even (i.e., proportional abundances are the same).

The Modified Simpson's Evenness Index is calculated as follows:

$$MSIEI = \frac{-\ln \sum_{i=1}^n P_i^2}{\ln n}$$

where P is the portion of soilscape occupied by a soil class i, and n is the total number of soil classes forming the soilscape. It is a composite index mathematically and equals the ratio of the MSIDI divided by the natural logarithm of the number of classes.

All the indices were computed in GIS environment using the FRAGSTATS tool (McGarigal et al. 2002). FRAGSTATS is a computer software program designed to compute a wide variety of landscape metrics that was implemented by ecologists to measure characteristics of landscapes and their components. One of the fashioning advantages of FRAGSTATS is that any calculation is fully integrated in a GIS and consequently easy to apply to digital maps (Raines 2002). Table 7.5 quantifies the changes of the soil classes from 1955 to 2050, while Fig. 7.3 shows the relative soil maps used to calculate pedodiversity indices in FRAGSTATS.

Richness index(s) is equal to 15 in 1955 and equal to 16 in the following years. As predicted for 2050(s) decrease to 13 as three soil classes, according the space-time prediction, would not be present any more (suborders: Typic Haploxerolls, Calcic Haploxerolls, and Pachic Haploxerolls). Diversity values (Shannon's, Simpson's, Modified Simpson's) (Fig. 7.10A) show a slight increase in pedodiversity from 1955 to 1966, a rapidly decreasing trend until 2008 and then a more marked decrease in the simulated 2050

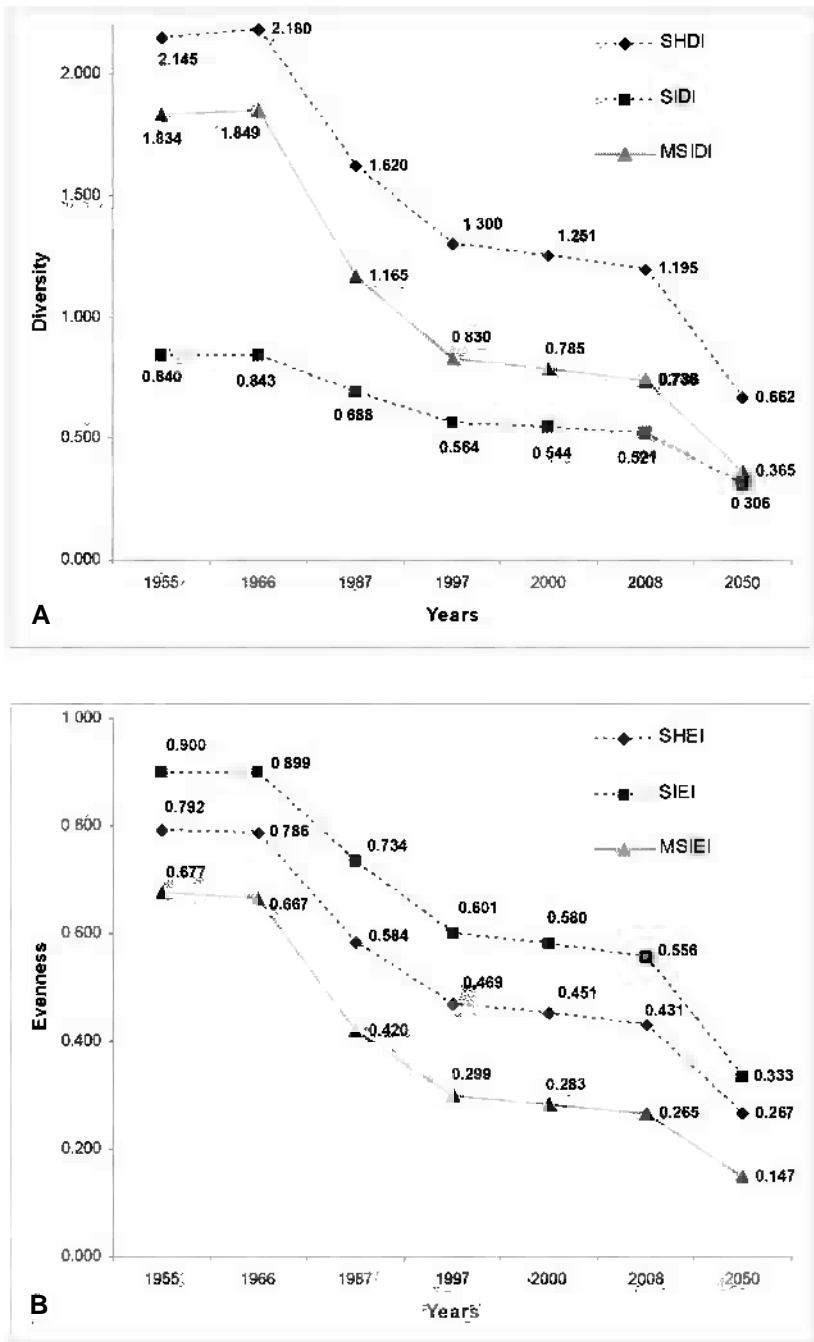


Figure 7.10 Pedodiversity indices calculated for every dated soil map in FRAGSTATS.

when three soil types disappear from the soilscape. Evenness indices (Fig. 7.10B) always show a decreasing trend in time, apart in first period of human intervention for the MSIEI, having the same values in 1955 and 1966, when anthropogenic soils were introduced in a small portion of the soilscape. Based on Simpson's index, we observe a relatively high value of evenness (0.9) indicating a very high distribution of area among classes; therefore, there is an even proportional contribution of each soil class in the system and there is not a dominance of any soil into the soilscape.

All the adopted diversity indices show a slight increase in the pedodiversity from 1955 to 1966, indicating that the introduction of the new soil type (anthropogenic soil) has resulted in higher variability and increasing diversity at soilscape level. In the specific case of Mazzarrone, considering what we observed in time, our indices clearly show that the human intervention in soil transformation could increase the diversity in the landscape in an initial phase, but subsequently, due to large scale farming, result in significant loss of diversity over time. Interpreting the diversity as explained by the evenness trend (always decreasing) we can also affirm that our soilscape was a stable system where the distribution of soil classes was in equilibrium. Soil transformation by large scale farming breaks the even equilibrium of Mazzarrone creating over time, a dominating soil class which makes the soilscape uniform and undermines the natural and innate soil variability of this area.

When considering the actual trend, we look at what is possible to foresee in the next 42 yr, i.e., in 2050, we can observe that the richness value will decrease from 16 to 13 with the complete disappearance of Typic, Calcic and Pachic Haploxerolls. The diversity indices and the evenness values will continue to decrease markedly (Fig. 7.10A and Fig. 7.10B).

Our model predicting the disappearance of soil types would be not far away from the real world if we take into account the soilscape change and its intensity from 1966 upto the present.

7. Spatial Variability of Some Soil Properties in an Anthropogenic Polypedon

To highlight how man can modify the natural configuration of the soils in producing an unpredictable chaos of the features of the soils, we have investigated the three-dimensional spatial variability of selected physico-chemical properties of Anthrosols from large scale farming (Palermo et al. 2009).

For the purpose of this survey, nine soil profiles were dug following a grid of 20 m inside a test area of 300 x 200 m, with a less than 2 meters difference in altitude along a North-Eastern direction. The soil profiles were

described and sampled following the ICOMANTH (2004) instructions. Soil samples were physico-chemically characterized. In particular we took into account the amount of clay and organic carbon, cation exchange capacity, pH and total and active carbonates. Data on soil properties were spatially analyzed to assess their variability in the three-dimensional space where soil profiles were located. The digital display of such volumetric space was performed by an irregular three-dimensional grid with the following step: $x = 1$ m; $y = 1$ m; $z = 0,025$ m. Each physical and chemical parameter was represented on the grid by a three-dimensional interpolation using the weighted method with the inverse of the square distance (IDW method), a very easy and time saving method.

The soil properties display consistent spatial variability. Total CaCO_3 (Fig. 7.11.1) increases along the East direction. Active CaCO_3 values (Fig. 7.11.2) are not in accordance with the total CaCO_3 due to the presence of added limestone material on the surface (Dazzi et al. 2009). The organic carbon trend follows the morphology of the field (Fig. 7.11.3). these trends are still visible at a depth of 80 cm, going from lower values (along 1-2-3 transect), to higher values (along 7-8-9 transect). Figure 7.11.4 shows a marked variability of the pH within the first 40 cm. The maximum and the minimum values (Table 7.7) are displayed along a direction that is parallel to that of the soil plowing. The CEC variability points out, more than the other soil parameters, how human intervention can alter the distribution of the physical and chemical soil characteristics.

In Fig. 7.11.5A it is clearly visible that the radial development of the CEC variability starting from $0,6 \text{ cmol}_{(+)} \text{ kg}^{-1}$ in profile 5, is quite different from the maximum value ($31,3 \text{ cmol}_{(+)} \text{ kg}^{-1}$).

Figure 7.11.6 show a decrease in clay content along the North direction at 100 cm depth and along the East direction at 20 cm depth, suggesting that the clay variability depends on the transformation of the pre-existing natural soils of the area which were Kastanozems.

8. Conclusions

As Targulian stresses (2008) soil responses to human impact depend not only on their intensity and duration but also on the direction of acting impacts: from the surface, laterally or from the bottom of the soil system. According to these impacts it is possible to discriminate three models of soil body responses (Targulian 2008): fan-like, when the uppermost soil horizons are changing stronger and faster than lower ones; belly-like, when the middle soil horizons are changing stronger and faster than upper and deeper ones; skirt-like, when the deepest soil horizons are changing stronger and faster than all upper occurring horizons.

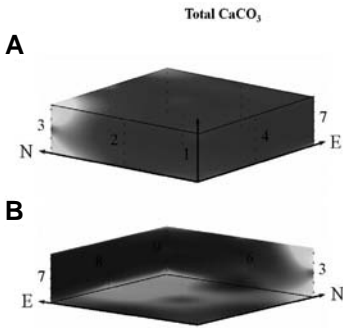


Figure 11.1

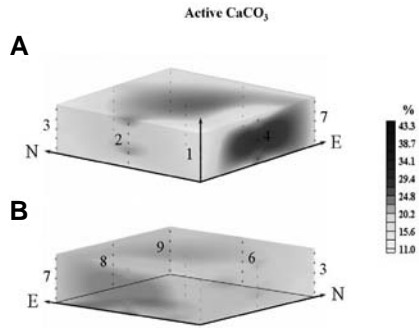


Figure 11.2

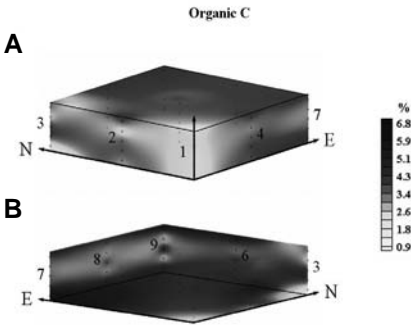


Figure 11.3

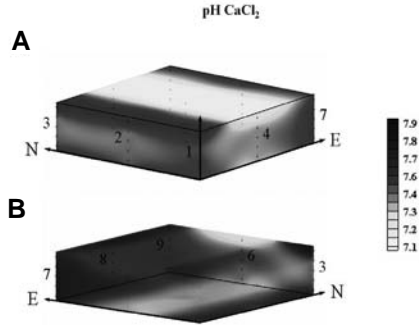


Figure 11.4

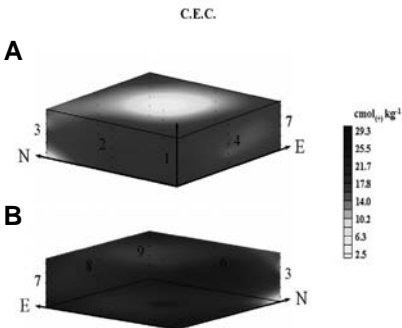


Figure 11.5

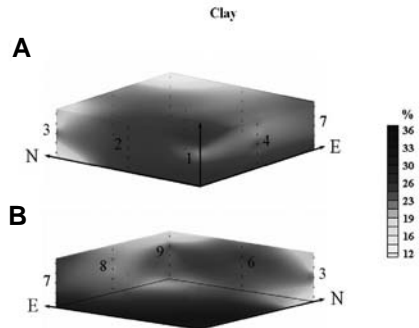


Figure 11.6

Figure 7.11.3 Dimensional spatial variability of the surveyed descriptors: A) top side view and B) bottom side view.

In our case man's action on soil cannot be ascribed to the three above mentioned models because his action on soils not only provides for the spreading of 50–70 cm deep calcareous human transported materials (Dazzi and Monteleone 2007) but also because of deep mixing of HTM (Dazzi et

Table 7.7 Descriptive statistics of the physico-chemical descriptors investigated.

profile	Total CaCO ₃ %				Active CaCO ₃ %				Organic C g kg ⁻¹			
	mean	st. dev.	min.	max.	mean	st. dev.	min.	max.	mean	st. dev.	min.	max.
1	58	±6	51	68	9,6	±1,9	7,5	12,5	1,5	±0,5	0,6	2,2
2	59	±6	48	64	18,8	±3,8	12,5	22,5	3,0	±1,5	0,6	4,3
3	37	±10	25	52	11,0	±2,8	7,5	15,0	3,0	±1,6	0,5	4,4
4	61	±14	32	69	29,0	±10,2	20,0	48,9	3,9	±0,7	2,6	4,5
5	52	±11	35	69	21,7	±2,0	20,0	25,0	3,6	±1,6	1,1	5,2
6	57	±9	45	71	18,1	±1,9	14,8	19,8	4,8	±1,4	3,1	7,2
7	65	±12	44	75	18,1	±0,7	16,8	18,7	4,3	±1,4	3,3	7,0
8	69	±15	44	82	18,1	±0,5	17,4	18,8	3,2	±2,5	1,9	5,5
9	61	±13	41	76	17,7	±0,6	16,5	18,5	4,0	±2,2	1,4	6,6
profile	pH (CaCl ₂)				C.E.C. cmol ₍₊₎ kg ⁻¹				Clay %			
	mean	st. dev.	min.	max.	mean	st. dev.	min.	max.	mean	st. dev.	min.	max.
1	7,6	±0,0	7,6	7,7	29,3	±2,8	24,4	32,5	31,5	±6,3	22,0	38,1
2	7,5	±0,0	7,5	7,5	27,1	±3,4	23,8	33,1	30,5	±4,5	27,2	39,3
3	7,5	±0,0	7,5	7,6	14,8	±4,3	8,7	20,0	17,3	±4,6	10,7	23,3
4	7,4	±0,0	7,4	7,5	22,2	±6,1	16,3	33,8	28,9	±8,9	21,3	45,4
5	7,4	±0,0	7,4	7,5	11,2	±7,9	0,6	21,3	26,7	±1,7	24,5	28,8
6	7,4	±0,0	7,3	7,5	25,0	±5,1	16,3	31,3	23,3	±1,2	22,2	25,6
7	7,8	±0,0	7,7	7,8	22,5	±3,0	20,0	27,3	27,4	±3,7	24,8	33,5
8	7,9	±0,1	7,8	7,9	20,3	±2,2	17,5	22,5	20,6	±1,7	18,5	22,6
9	7,9	±0,0	7,8	7,9	19,2	±3,9	13,8	23,8	20,5	±3,9	11,7	24,0

al. 2009). This is the reason why we propose a fourth model of soil body response to the human impact: the “millwheel” response whereby all the soil horizons are mixed deeper, stronger and faster (also by adding HTM) than is possible to identify any remnants of the previous natural soil horizons. In these soils there is no logical distribution of organic or mineral elements and any connection between them, and the endogenic fluxes of energy into the soil are completely lost.

Many times, man’s activity as a soil forming factor can be traced back to agricultural land management. In our study case, the farmers’ observation—during the 1970’s—that grapes could be grown in a very suitable environment was the motivating factor of large scale land-use changes that involved the soilscape in a very severe way mainly on the basis of economics. The main changes took place in converting some land-use classes towards trellis system vineyards. This particular land use causes great modification in the environment that is very visible in the landscape as well as in the soilscape. The social and economic benefits in Mazzarrone were obtained through a deep action over the soils. Farmers, in most cases, are interested in increasing economic profits, and the higher they accrue, the more they exploit natural resources, in particular the soil. These changes are wrought without considering that each soil has its own evolutionary process and its own activity that contributes to pedodiversity.

The land in Mazzarrone is a very clear example showing the effects of the land-use change on the soilscape by large scale farming over time. The rapid expansion of vineyards for grape production involved deep irreversible transformation of the soils leading to the creation of new anthropogenic soils. At present, these soils represent the dominant type in the Mazzarrone soilscape. One of the effects of the expansion of the anthropogenic soils was to decrease pedodiversity. The pedodiversity indices, both in terms of diversity and evenness, showed unequivocally a substantial decrease over time. The simulated expansion in a future scenario indicates that pedodiversity would continue to decrease reaching very low values, highlighting the possibility that some original soil types could be completely transformed.

The loss of pedodiversity is in itself considered as a negative phenomenon disturbing the soil ecosystem that is a fundamental component of planetary life. However, in the case of Mazzarrone, and wherever pedodiversity is affected by this kind of land-use change, many other negative environmental impacts could also be present (erosion, loss of biodiversity, pollution, etc.).

Man’s action in large scale farming aiming at obtaining soils with a high degree of suitability for vineyards cultivation, even if it can be considered as a methodology of pedo-technique, leads to a peculiar heterogeneity of the physical and chemical parameters of the anthropogenic soil, with respect

to natural soils. From a pedogenetic point of view, man's action in creating anthropogenic soils can be seen as an involuntary action that favors a strong heterogeneity of soil features and that increases disorder with the soil as well as a decrease in pedodiversity.

The prediction of what can reasonably happen in the future allows for a better comprehension of the phenomena that are linked to land-use change and should be used to stimulate local stakeholders and land managers towards sustainable land use and the protection of the pedological heritage.

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CHAPTER 8

Soil Endemism and its Importance to Taxonomic Pedodiversity

James G. Bockheim* and Nicholas Haus

1. Evolution of the Concept of Soil Endemism

Endemism is a key concept in ecology and refers to naturally occurring plant or animal species that are confined to a particular geographic area (Magurran 1988). Endemism may occur at any scale and normally involves isolation of individual species, groups of species or entire communities. Isolation may be due to displacement by continental drift, island building, long-term stability of landforms and other causes.

Biogeographers recognize three kinds of endemism: (1) edaphic endemism refers to species that have a strong fidelity to a specific soil taxon and is found nowhere else (e.g., Whittaker et al. 1954), (2) neoendemism refers to species that have speciated relatively recently, and (3) paleoendemism refers to species with a more ancient lineage (Kruckeberg and Rabinowitz 1985). Kruckeberg and Rabinowitz (1985) defined *narrowly endemic* taxa as those that occur in one or a few small populations and are, therefore, confined to a single location.

Guo et al. (2003a) may have been the first to use the term “soil endemism,” defining it as “the taxa-richness of soil communities”. Amundson et al. (2003) observed that the overall diversity and soil endemism in California

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could be understood in terms of the wide range and unique combinations of climate, vegetation, and geology within the state. Guo et al. (2003b) related the concept of endemism to the taxonomic level of soil series.

Bockheim (2005) raised the question as to whether or not the term “endemism” applies to soils. There are several arguments favoring the use of the term endemism with regard to soils. Soil-formation theory strongly implies that endemism should occur in soils. Dokuchaev (1899) recognized that the array (zonation) of broad soil taxa is governed primarily by climate and its influence on regional vegetation. The soil-forming factor equation, first enunciated by Shaw (1932) and later developed by Jenny (1941), suggests that a soil body is a product of the integrated effects of climate, organisms, relief, and parent material all operating over time. Therefore, unique soils should occur in areas reflecting a unique combination of soil forming factors, and areas with a similar combination of soil-forming factors should contain similar soils. For example, the soil catena, which is an interlocking of soils on a landscape (Milne 1935, Bushnell 1942), features a repetitive pattern of soil series on the landscape in relation to aspect and slope position. “The unique combination of state factor combinations from region to region would be expected to result in unique soils, and the number of state factor combinations would be expected to increase with increasing land area” (Guo et al. 2003b).

Secondly, soils commonly have an “isolating mechanism” that causes its populations to separate. The cases mentioned in the next section with regards to “edaphic endemism” are examples of the geographic isolation that contributes to soil endemism. Geographic centers of plant diversity and endemism have been identified throughout the world. For example, areas identified by the North American Regional Centre of Endemism (2004) of high plant diversity may be correlative with areas of high pedodiversity (Guo et al. 2003a,b). Paleoendemism and neoendemism are examples of temporally isolated landforms. Recently deglaciated or volcanic regions are representative of neoendemism, and geomorphically stable landforms such as ancient stream terraces exemplify paleoendemism.

A third argument favoring the concept of soil endemism is that of “soil memory.” Targulian and Goryachkin (2004) defined soil memory as the capacity that soil systems have for storing information about environmental factors and pedogenic processes that have been acting during pedogenesis. In other words, soil memory is the unique set of soil properties inherited by the interactions of the soil-forming factors. They identified the main mechanisms of soil memory as “those sets of pedogenic processes that generate the solid phase products and features within the multiphase soil system”. The main types of solid-phase carriers of soil memory and their spatial/temporal hierarchy within the soil system were briefly described. They emphasized the importance of isomorphism and polymorphism of soil-

record carriers with regards to pedogenic processes in attempting to decode and understand the information stored in soil memory. When reading the record in the soil system under one type of climate, it is obligatory to account for the diversity of solid phase soil horizons and pedons induced by the diversity of parent materials (lithodiversity), topography (topodiversity), biota (biodiversity) and duration of pedogenesis (chronodiversity). An endemic soil's memory, therefore, can be expected to retain the unique set of pedoevolutionary properties similar to endemic plant and animal species.

It could be argued that soils do not feature endemism in the strictest sense because they are incapable of migrating. Moreover, the soil is not an organism that has the capacity to reproduce or that is subject to behavioral isolation (e.g., as when a population develops two different mating seasons or behavior). Rather, the soil is a "bioreactor" (Targulian and Sokolova 1996) that integrates all of the soil-forming factors and the temporal and spatial changes in these factors.

Geostatistics may also argue against the concept of soil endemism. McBratney (1992) emphasized the uncertainty in predicting the spatial distribution of soils on the basis of statistics, simulation modeling and multivariate techniques. At a continental level, pedorichness and pedodiversity are characterized by similarities rather than differences (Ibáñez et al. 1998). However, if soil taxa are geographically restricted to one or a few locations or regions, endemism applies. This is especially true for different climatic regimes. For instance, pedodiversity is higher in the mid-latitudes than in the polar regions (Ibáñez et al. 1998).

2. Examples of Edaphic Endemism

The concept of edaphic endemism refers to soil parent materials that either have toxic quantities of certain chemical elements or are so deficient in plant nutrients that only certain plants can survive on them. Excellent reviews of edaphic endemism were given by Kruckeberg and Rabinowitz (1985), Kruckeberg (1986), Cowling et al. (1994), and Rajakaruna (2004). There are numerous examples of edaphic plant endemism from the literature (Table 8.1). Perhaps the most commonly cited example is the influence of ultramafic conditions on plant endemism (Whittaker et al. 1954, Kruckeberg 1986). Ultramafic parent materials such as serpentinite contain high levels of Mg, Cr and Ni often at the expense of other elements such as Ca. Serpentine areas usually support a stunted type of vegetation that is unique in its species composition. In contrast to ultramafic soils, soils derived from "chalk barrens" or gypsum support endemic plant species because of physical soil properties such as water availability or bulk density rather than nutrient imbalances (Meyer 1986, Cowling et

Table 8.1 Examples of edaphic plant endemism from the literature.

Soil parent material	Locations	Citation
Ultramafics (serpentinite)	Siskiyou Mtns., OR; Klamath Mtns., CA; South Africa; Italy, Balkan Peninsula; New Caledonia; Zimbabwe; Yugoslavia	Whittaker et al. 1954, Kruckeberg 1986
Xeric limestone prairie/ chalk barrens; calcareous sands	MO; CA; CO; Australia; South Africa	Cowling et al. 1994, Rajakaruna 2004
Siliceous sand	Amazonian Peru; Australia; South Africa	Cowling et al. 1994
Gypsum	CA; central Mexico; central Spain	Meyer 1986
Mine tailings/heavy metals (Cu, Zn)	Europe	Rajakaruna 2004
Heathlands	w. Mediterranean Basin; South Africa	Rodriguez-Sanchez et al. 2008, Prance 1996
Inselbergs	South Africa; Brazil	Prance 1996
Unglaciaded regions	South Africa; Australia	Cowling et al. 1994, Prance 1996
Mangroves	[general]	Kruckeberg and Rabinowitz 1985
Submerged aquatic	[general]	Kruckeberg and Rabinowitz 1985
Rock outcrops/coastal bluffs	CA	Rajakaruna 2004

al. 1994). Endemic plants also are common in areas such as South Africa and Australia where soils are strongly weathered and of great antiquity (Cowling et al. 1994, Prance 1996).

3. Measures of Soil Endemism

Although they are widely used in biology, indices of endemism do not appear to have been applied in pedology. Quantitative methods of endemism should account for the spatial diversity of a soil body at a defined level of precision, which is normally based on a regular lattice or grid, latitudinal range, or for a defined region divided into cells and occasionally for “hotspots” of restricted area. The spatial resolution is defined by the cell size (Laffan and Crisp 2003).

Five indices of endemism have been used in biology, including (i) simple endemism, (ii) threshold endemism, (iii) weighted endemism, (iv) corrected-weighted endemism (CWE), and (v) endemism-richness (Table 8.2). Simple endemism (Eq. 1) refers to the percentage of a soil body

confined to a mapping unit. The disadvantages of using this index are that it does not (i) tell about range sizes of soil bodies not confined to the mapping unit or (ii) take into account the size of the mapping unit.

Threshold endemism (Eq. 2, Table 8.2) is based on the upper range limit, i.e., the number of range-restricted soil bodies in a grid cell or mapping unit. According to Crisp et al. (2001), the main disadvantage of this measure pertains to subjectivity in assigning scores to range size classes.

Weighted endemism (Eq. 3, Table 8.2) uses a continuous weighting function. For example, species (or soil bodies) with small ranges are assigned higher weights, and those with larger ranges are given lower weights. The weighted index is roughly proportional to the reciprocal of a species' range. The disadvantage of this index is in selecting a meaningful functional form. In corrected weighted endemism (Eq. 4, Table 8.2) values are adjusted so that they range between 0 and 1, and the technique records the average per species range restricted to a cell. CWE is the inverse average number of localities or habitats occupied by each soil body within the larger region (Laffan and Crisp 2003).

Table 8.2 Indices of endemism.

Name	Equation	Parameters	Reference
Threshold endemism	$TE = I_i / (I_i + E_i) * 100$	I_i = range size of species within the mapping unit; E_i = range size of species I outside the mapping unit	Kier and Barthlott 2001
Weighted endemism	n $WE = \sum G_i^{-1}$ $i = 1$	n = no. of species per study group; G_i = global range size	Linder 2001, Kier and Barthlott 2001
Corrected-weighted endemism (beta diversity)	$CWE = WE / n$	WE = weighted endemism; n = total no. of species in a grid cell	Crisp et al. 2001, Linder 2001
Endemism index	n $E = (\sum G_i^{-1}) n^{-1}$ $i = 1$	G_i = range size of species i measured as the number of cells covered by the species' global range; n = no. species per study group	Kessler 2001
Proportional endemism (endemism richness)	n $E = \sum G_{i(e)} / G_{i(t)}$ $i = 1$	n = no. of species per study group; $G_{i(e)}$ = no. of species of group i in study region; $G_{i(t)}$ = total no. of species of group i	Kier and Barthlott 2001, Lamoreaux et al. 2006

Finally, endemism richness (Eq. 5, Table 8.2) refers to the total number of soil bodies per cell or ecoregion, divided by the total number of soil map units in the database for that group (Kier and Barthlott 2001, Lamoreaux et al. 2006). This index allows calculations to be comparable between taxa without a single species group overwhelming the others.

4. Procedure for Identifying Endemic Soils

This study uses the USDA Natural Resources Conservation Service (NRCS) database and three approaches for the identification of endemic soils in Wisconsin. In the first approach, the number of soil orders, suborders, great groups, subgroups, families, series, phases, and map polygons were determined for each of Wisconsin's 72 counties using the NRCS "Geospatial Data Gateway" and county scale data contained in SSURGO (<http://soils.usda.gov/survey/geography/ssurgo/description.html>) (Soil Survey Staff 2012a).

The second approach focused on narrowly endemic soils in Wisconsin. The ranges of each of the 830 soil series listed in the Soil Series Classification Database for Wisconsin (Soil Survey Staff 2012b) were examined. Soil series with a centralized distribution, regardless of total area, were selected for analysis. Soil series with disjunct populations, i.e., populations more than one county or more than one state away, were used on a few occasions and only if these populations had similar climatic, ecologic, or physiographic conditions (homoclimes) in their different locations (primarily within Wisconsin). Soils with a broad range that transgressed homoclimes were eliminated as not being narrowly endemic. Official Soil Series Descriptions (OSD) (Soil Survey Staff 2012c) were examined to assure that there were no competing soils, i.e., soils in the same family. In several cases, competing soils were utilized if they were sufficiently different, despite being in the same family. A spreadsheet of soil series considered as unique and endemic, with their areas and taxa by level, was prepared. The series were then examined to see what properties appeared to contribute to their endemism. The approach is adapted from Linder (2001) and summarized as follows:

- The soil series is defined as a "species."
- Each area, defined here as a county, must have at least two endemic soil species.
- The ranges of the soil species endemic to the area of endemism should be maximally congruent.
- The areas of endemism should be narrower than the whole study area.
- Areas of endemism must be mutually exclusive and grid squares (or any other units) cannot belong to two areas of endemism.

Especial attention was given to the environmental tension zone in Wisconsin, which constitutes a transition between the central hardwood forest-prairie to the south and the northern mixed hardwood and conifer forest to the north. The WTZ contains an unusually large number of plant species at their southern or northern range limits (Curtis 1959) and is also a transitional zone for animals, such as insects, birds and other animals. The WTZ follows the mean July air temperature isotherm of 21°C.

The approach used for Wisconsin was also used to identify the proportion of endemic soils in several other states based on their land area and their number of Major Land Resource Areas (MLRAs) (USDA NRCS 2006). Major land resource areas, which total 278 in the USA and its territories, are geographically associated land resource units. Identification of these large areas is important in statewide agricultural planning and has value in interstate, regional and national planning.

The third approach entailed comparing the frequency distribution of endemic soils in Wisconsin to that of endemic vascular plants. To achieve this, we utilized databases from the Department of Botany at the University of Wisconsin (<http://botany.wisc.edu/cgi-bin/GenusList.cgi>) and the Natural Heritage Inventory (NHI) from the Wisconsin Department of Natural Resources (<http://dnr.wi.gov/org/land/er/biodiversity>). The NHI shows the distribution of endemic species on range maps. We determined the number of endemic vascular plants for each county in Wisconsin and compared that to the number of soil series in each county using linear regression.

5. Endemic Soils in Wisconsin

Endemic soils in WI occur in 7 orders, 14 suborders, 29 great groups, 83 subgroups, 138 families, and 159 soil series (Table 8.3), comprising 19 percent of the total soil cover in the state and an area of 1.44 million ha. There is a significant correlation between the log of the number of endemic soil taxa in Wisconsin and the log of hierarchical rank (Fig. 8.1). These findings are similar to those found for pedodiversity (Ibáñez et al. 2009). For USA soils as the taxonomic category decreased from order to series, Shannon's diversity index increased, because taxa richness increased dramatically (Guo et al. 2003b). Similarly, the greatest number of endemic plants occurs at the family level or below (Major 1988).

There is a high correlation between soil richness (pedodiversity) and soil endemism (Fig. 8.2), a finding that has also been reported for plants. The predominant soil properties related to soil richness and soil endemism in Wisconsin include (i) the presence of a glossic horizon (34 percent of endemic soils), (ii) aquic conditions or oxyaquic subgroups (28 percent), and (iii) bisequal profiles (20 percent of endemic soils) (Table 8.4). The

Table 8.3 Endemic soils by taxonomic level in Wisconsin.

Orders (7)	Suborders (14)	Great groups (29)	Subgroups (83)	Area (ha)	Families ¹ (138)	Series (159)
Spodosols	Orthods	Haplorthods	Alfic, Alfic Oxyaquic, Entic, Entic Lithic, Fragic,	203715	13	Stambaugh, Vanzile, Superior, Abbaye, Fence, Lapoin, Guenther, Ashwabab, Eauclaire, Metonga
			Oxyaquic Ultic			Arbutus, Rockdam, Redrim, Michigamme, Humbird, Ludington
		Fragiorthods	Alfic, Alfic Oxyaquic, Oxyaquic	270810	5	Laona, Schweitzer, Wakefield, Gogebic, Argonne, Wabeno, Champion
	Aquods	Endoaquods	Argic, Typic	42329	3	Whisklake, Channing, Monico, Ironrun
		Epiaquods	Alfic, Ultic, Typic	60104	5	Sedgwick, Lablatz, Wayka, Merrilan, Fairchild
		Fragiaquods	Argic	18304	1	Tula
Alfisols	Udalfs	Hapludalfs	Aquic Arenic, Aquollic, Arenic, Arenic, Oxyaquic, Lamellic, Lithic, Mollic, Mollic Oxyaquic, Typic	79100	18	Wyeville, Meenon, Stengel, Zittau, Rockmarsh, Neenah, Perida, Fremstadt, Karlsborg, Pearl, Brice, Muscoda, Norgo, Borth, Kaukauna, Dairyland, Winneconne, Omro, Kranski, Bigisland, Puchyan
		Glossudalfs	Aquertic, Aquic, Arenic, Haplic, Oxyaquic, Oxyaquic Vertic, Typic, Vertic	389868	19	Taylor, Ossmer, Spear, Siren, Vlasaty, Poskin, Almena, Comstock, Maplehurst, Moshawquit, Rabe, Antigo, Anigon, Brill, Denomie, Tradelake, Billyboy, Brander, Amnicon, Anton, Frechette, Alban, Miskoaki
		Paleudalfs	Aquic, Mollic, Oxyaquic	111797	4	Reedsburg, Wildale, Valton, Brinkman, Haugen
	Aqualfs	Albaqualfs	Arenic	350	1	Dody
		Endoaqualfs	Aeric, Mollic, Udollic	46526	4	Banat, Annriver, Mequon, Sherry

		Epiaqualfs	Chromic Vertic, Mollic, Typic, Umbric, Vertic	113925	6	Alango, Marshfield, Wozny, Bjorkland, Chelmo, Indus, Auburndale, Barronett
		Glossaqualfs	Aeric, Mollic	111531	5	Dolph, Oronto, Borea, Cuttre, Cebana, Glenflora
Ultisols	Udults	Hapludults	Aquic	244	1	Sioux creek
Inceptisols	Udepts	Dystrudepts	Aquic, Aquic Humic, Humic Psammentic, Lithic, Oxyaquic, Typic	55866	8	Mylrea, Sechler, Impact, Mindoro, Haustrup, Slimlake, Spoonerhill, Moberg, Elderon, Keyesville, Mosinee
		Eutrudepts	Aquic Dystric, Oxyaquic	14358	4	Clemens, Nebago, Fisk, Nichols
	Aquepts	Humaquepts	Histic	20706	2	Makwa, Wildwood
		Endoaquepts	Mollic	8135	1	Lows
		Epiaquepts	Humic, Vertic	56668	3	Vesper, Veedum, Lerch
Histosols	Saprists	Haplosaprists	Terric	19301	2	Dawsil, Adder
Entisols	Aquents	Epiaquents	Humaqueptic, Mollic	32375	2	Elm Lake, Wautoma
		Fluvaquents	Mollic, Typic	14081	2	Waupaca, Totagatic
		Psammaquents	Humaqueptic	16110	2	Ponycreek, Newlang
	Psamments	Quartzipsamments	Aquic, Lamellic, Oxyaquic, Typic	11821	4	Majik, Windward, Tintson, Twinmound, Simescreek
	Orthents	Udorthents	Oxyaquic	238	1	Skog
	Fluvents	Udifluvents	Aquic, Mollic, Typic	10353	4	Wega, Dechamps, Boguscreek, Absco
Mollisols	Aquolls	Epiaquolls	Typic	46267	2	Poygan, Mann
		Endoaquolls	Lithic, Typic	14343	4	Ruse, Beartree, Warman, Shag
	Udolls	Argiudolls	Aquic, Oxyaquic, Typic	11756	7	Smestad, Hoop, Freya, Montello, Lara, Peebles, Nickin
		Hapludolls	Aquic, Fluventic, Lithic, Typic	7769	5	Soderbeck, Quarderer, Drylanding, Dunnville, Garne

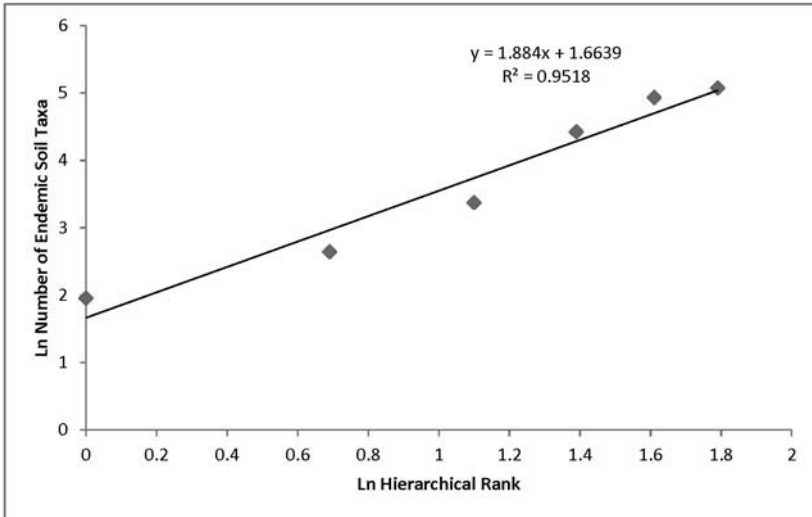


Figure 8.1 Relation between log of number of endemic soil taxa in Wisconsin and log of hierarchical rank.

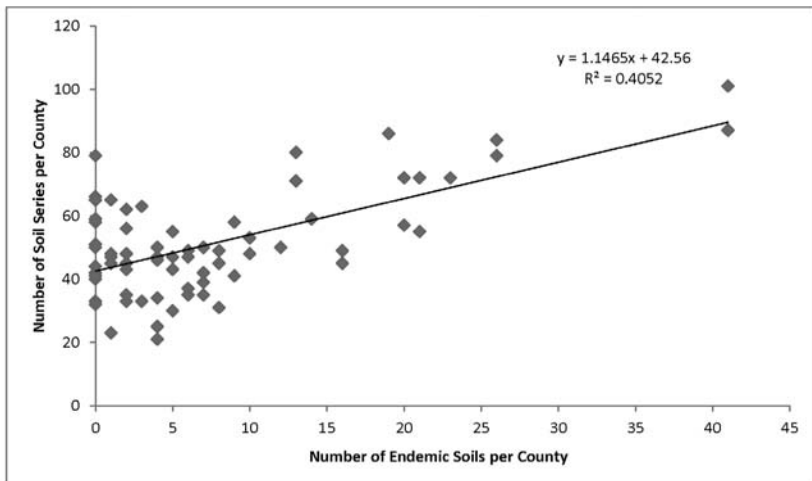


Figure 8.2 Relation between number of soil series per county and number of endemic soils per county in Wisconsin, USA.

ecological tension zone (TZ) that extends southeast to northwest across the state contains an unusually rich pedodiversity, including 6 of the 7 soil orders delineated in the state, 10 of 20 suborders, 17 of 50 great groups, and 28 of about 200 subgroups (Fig. 8.3).

Table 8.4 Soil properties associated with endemic soils in Wisconsin.

Factor	No. of soil series	% of total	Area (ha)	% of total
Climate				
Wet (SPD, PD, VPD, MWD) ¹	58	36.5	396284	27.5
Parent material				
Siliceous mineral class	6	3.8	35289	2.4
Lithic or paralithic contact	13	8.2	88533	6.1
Lithologic discontinuity (not bisequal)	12	7.5	31733	2.2
Complex parent material	8	5.0	39434	2.7
Environmental change				
Glossic horizon	35	22.0	484418	33.6
Bisequal	24	15.1	290412	20.1
Time				
Paleo B	3	1.9	75359	5.2
Total	159	100	1441462	100

¹SPD = somewhat poorly drained; PD = poorly drained; VPD = very poorly drained; MWD = moderately well drained.

There are 102 vascular plant species endemic to WI, the distribution of which is related more to specific habitat than to soil taxa and includes cliffs (20 percent of endemic plants), prairies (20 percent), emergent aquatic environments (13 percent), and sand dunes (12 percent of endemic plants). On a county basis, there is a poor correlation (adjusted $R^2 = 2.2\%$, $p = 0.11$) between the number of endemic vascular plant species and the number of endemic soil series. These findings contrast with those from other studies. For example, Cowling et al. (1994) reported that 90 percent of the endemic plants in Australia were “edaphic specialists.” However, as Kruckeberg and Rabinowitz (1986) point out, plant endemism is controlled by organism interactions as well as by soil climate and parent materials.

6. Endemic Soils from other States in the USA

We observed that states vary in the number of soil series (i.e, species richness) and the proportion of soil series that are endemic to the state. More interestingly, there was a strong relation between the proportion of endemic soil series, the land area of the state, and the variability in physiography as represented by the number of Major Land Resource Areas (MLRAs) (Fig. 8.4). For example, large, diverse states such Alaska and California have a larger proportion of endemic soils (27–33 percent) than smaller and less physiographic diverse states such as Rhode Island and Indiana (7 percent).

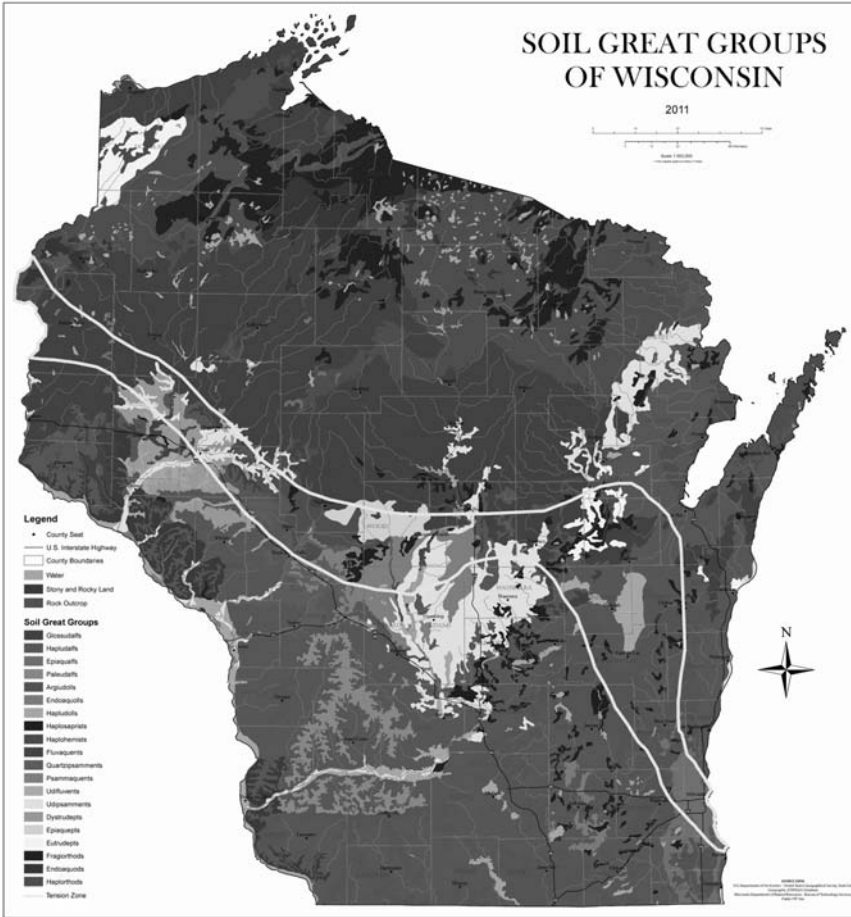


Figure 8.3 Soil great-group map of Wisconsin showing the ecological tension zone. Color image of this figure appears in the color plate section at the end of the book.

7. Soil Endemism and Ecosystem Management

The of endemism is important for ecosystem management, including the identification and protection of rare and endangered soils (Amundson et al. 2003, Drohan and Farnham 2006, Bockheim 2010). Amundson et al. (2003) reported 4.5 percent of the soils in the USA to be in danger of substantial loss, or complete extinction, due to agriculture and urbanization. Kulmatinski et al. (2004) recognized the ability of endemic soils to control invasive weeds in wheatlands of eastern suggesting that endemic soil-

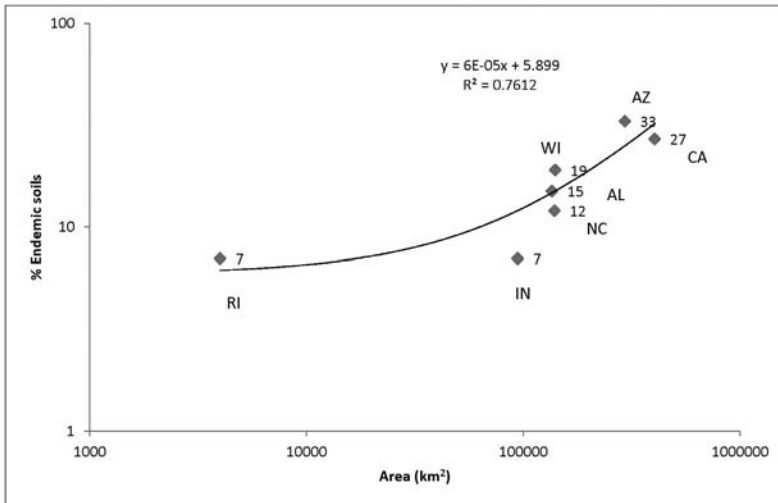


Figure 8.4 Relation between the percentage of endemic soils by state and (A) state land area and (B) the number of Major Land Resource Areas in the state.

based mechanisms of weed control may be common and may yield novel techniques for the management of invasive plants.

“Terroir” is a concept in viticulture that relates the sensory attributes of wine to the environmental conditions in which the grapes are grown. For the production of high quality red wines, environmental conditions should induce moderate vine vigor, either through moderate water deficit stress or through low nitrogen supply, conditions most frequently met on shallow or stony soils, in moderately dry climates (van Leeuwen and Seguin 2006). These findings suggest that endemic soils may be sought after for specialized landuse.

Worldwide pressure on soil resources has led to the diminished areas of minimally disturbed soils (Guo et al. 2003b). Recently, there has been interest in “rare and endangered” soils (Drohan and Farnham 2006). Ditzler (2003) suggested that rare soils be recognized as those with a mapped area of less than 10,000 ha and that unique soils be those that occur in only one state. He further proposed that endangered soils be those rare and unique soils that have had more than 50 percent of their area impacted by urban or agricultural development. Drohan and Farnham (2006) prepared a provisional list of soils that were rare or threatened in the US based on the following factors: (i) economic, (ii) ecologic, (iii) scientific, (iv) historic or cultural, and (v) rarity.

8. Conclusions

We consider endemic soils to be those that are found in narrow geographic areas as a consequence of a unique interaction of soil-forming factors. The concept of “edaphic endemism” recognizes that soils are important relative to endemic species of plants and animals. There are five measures of endemism widely used in biology; however, these measures do not appear to have been applied in pedology. A protocol is proposed for identifying endemic soils using the USDA Natural Resources Conservation Service soils database. The approach was used to delineate endemic soils in Wisconsin and several other states in the USA.

From the present study, the following conclusions may be drawn:

- Unlike plants and animals, soils are “plastic” in their response to environmental change, i.e., they have a “memory”;
- Endemic soils are most common at the lowest categories, e.g., families and series in *Soil Taxonomy*;
- There is a high correlation between soil richness (pedodiversity) and soil endemism; and
- The correlation between soil endemism and plant endemism is strongly dependent on geographic region.
- Large and physiographically complex states have the greatest proportions of endemic soils.

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CHAPTER 9

Pedodiversity Studies in China

Xuelei ZHANG

1. Introduction: Initiation of Pedodiversity Research in China

Pedodiversity describes soil variety in a particular area systemically, and this variety has a direct or indirect relation with soil types and soil conditions. In recent years, pedodiversity has become an important component of pedological studies (Ibáñez et al. 1995a,b, 1998, Ibáñez and De-Alba 1999, Phillips and Marion 2005, 2007, Ibáñez et al. 2005a, Saldaña and Ibáñez 2007). Toomanian et al. (2006) studied the forms and processes of landscape development and their subdivision in the main region of Zayandeh-rud Valley in Iran. They measured the degree of soil heterogeneity, reporting that Shannon entropy index was a good measure of soil evolution. Minasny et al. (2010) introduced the concept of taxonomic distance to pedodiversity by deriving taxonomic distances in the World Reference Base for Soil Resources (WRB), and compared traditional diversity measures based on abundance of soil individuals with those containing taxonomic distance.

After a review of research practices on pedodiversity in Europe and America, a Chinese research group led by Xuelei Zhang visited the Centro de Ciencias Medioambientales (CCMA) in September 2002 to learn how to apply the ideas used in Spain by Ibáñez and his team to China. Key research questions were how to (i) use and improve soil databases, (ii) monitor changes in pedodiversity in response to the changing land-use conditions, (iii) establish the societal value of undisturbed rare and unique soils, and (iv) address pedodiversity conservation (Zhang et al. 2003a,b, 2004a,b).

2. Background: Support of the National Natural Science Foundation of China

Since the theory and methodology of pedodiversity studies were introduced from abroad into China at the beginning of this century, three main steps have been taken including: (i) development of the basic tools for calculating spatial variation in Shandong and Hainan provinces, (ii) determination of the changes in pedodiversity from land-use changes in the Yangtze delta, and (iii) identification of the impact of intensive human activities under the fast growing urbanization upon regional pedodiversity patterns and the related soil resource implication. The National Natural Science Foundation of China (NSFC) has provided four rounds of funding to support research on pedodiversity, the most recent one to study the impact of land-use change on pedodiversity in China. This program is using the most advanced pedodiversity techniques in cooperation with international colleagues to quantify pedodiversity patterns with changes in land-use in the most developed and urbanized areas of China, including the eastern Yangtze delta and agricultural areas of central China. The approach being utilized involves the determination of pedodiversity variation and use of a connection index tool to illustrate the impact of changing land-use on regional pedodiversity. A key goal is to improve the sustainable use of regional soil resources and develop a healthy environment.

3. Main Research Progress Since 2001

3.1 Soil spatial variation analysis in Shandong and Hainan using SOTER

During the years 2001–2004, soil spatial variation was analyzed in Shandong (Tan et al. 2003) and Hainan (Zhang et al. 2001, 2003a,b,c, 2004a,b,c,d) provinces in the east and south China by using SOTER (Soil and Terrain Digital Databases). The 1:1 000 000 scale SOTER databases of Shandong province includes digital map units and their related attribute data on lithology, terrain and soil types. The attributed data are managed by a relation database manager system (RDBMS) and can be acquired and linked by their unique identification numbers to digital map units using GIS software. Richness index, evenness index and diversity index were selected as indicators of pedodiversity analysis. The Pielou index (E) (Pielou 1975) was used as evenness index and the Shannon index (H') (Shannon and Weaver 1948) as a diversity index. The pedodiversity indices were determined for soil groups derived from different types of parent materials and distributed on different terrains (Tables 9.1 and 9.2).

Table 9.1 Pedodiversity indices of soil groups derived from different types of parent materials in Shandong province.

Parent material	Symbol	Area	Pedodiversity index		
			S	H'	E
		km ²			
Igneous rock	I	40634	5	1.02	0.64
Acidic igneous rock	IA	32045	4	1.00	0.72
Intermediate igneous rock	II	6927	3	0.41	0.37
Ultrabasic igneous rock	IB	1662	3	1.00	0.91
Metamorphic rock	M	26570	6	0.89	0.53
Acidic metamorphic rock	MA	26570	6	0.89	0.53
Sedimentary rock	S	32434	5	0.97	0.60
Clastic sediment	SC	9551	4	0.93	0.67
Organic sedimentary rock	SO	22883	4	0.91	0.66
Unconsolidated material	U	51191	6	0.49	0.27
Unconsolidated-fluvial material	UF	51191	6	0.49	0.27

Table 9.2 Pedodiversity indices of soil groups developed on different terrains in Shandong province.

Terrain	Symbol	Area	Pedodiversity index		
			S	H'	E
		km ²			
Level land	L	101209	11	1.72	0.72
Plain	LP	77141	10	1.75	0.76
Depression	LD	20367	3	0.28	0.26
Valley floor	LF	3701	3	0.82	0.74
Sloping land	S	47531	8	0.86	0.42
Medium-gradient hill	SH	12854	5	1.00	0.62
Medium-gradient mountain	SM	4003	4	1.16	0.84
Medium-gradient escarpment zone	SE	30674	3	0.30	0.27
Steep land	T	2084	1	0	-
High-gradient mountain	TM	2084	1	0	-

S: Richness index, H': Shannon index and E: Pielou evenness index

3.1.1 Richness

The highest richness indices ($S = 6$) of soils were obtained from the largest soil-map units, acidic metamorphic rock (26 570 km²) and unconsolidated-fluvial material (51,191 km²); in contrast, the smallest areas, intermediate and ultrabasic igneous rocks, had the lowest richness indices ($S = 3$) (Table

9.1). These findings conform to those of Ibáñez et al. (1995a, 1998) and Ibáñez and De-Alba (1999) that the richness index increases with the area of the soil-map unit. The correlation coefficient between soil richness index and the soil areas for the different parent materials was 0.804 (significant at the 0.01 level). These findings imply that more sub-classes of acidic metamorphic rock should be delineated. There are mainly three kinds of landforms in Shandong province, and their richness indices were in the following sequence: level land (101 209 km²) > sloping land (47,531 km²) > steep land (2 084 km²) (11, 8 and 1, respectively) (Table 9.2). As with soil parent materials, the richness index increased with the area of soil-terrain units. The richness index could reflect the maturity of soilscales. Soil development is more favourable on flat landforms, whereas sloping and steep lands are easily eroded and are not stable for soils to develop fully. The correlation coefficient between richness index and the area of soils developed on different landforms was 0.862 (significant at the 0.01 level).

3.1.2 Diversity

Taking the igneous rock, metamorphic rock, sedimentary rock and unconsolidated material as examples, the Shannon diversity index (H') of soils derived from these four kinds of parent materials followed the order: igneous rock > sedimentary rock > metamorphic rock > unconsolidated material (Table 9.1). As mentioned previously, richness indices of metamorphic rock and unconsolidated materials were larger than those for soils developed on sedimentary and igneous rocks. Consequently, differences could be seen between diversity as well as richness of soils derived from different parent materials. Diversity takes into consideration not only their individual numbers (richness index) but also the equitability of objects in the system (evenness index) (Usher 1983). When measuring pedodiversity in a large-scale area, the diversity and richness indices always show the same trend. Moreover, the diversity evenness indices showed a similar trend in their relationship with soil area. The correlation coefficient was 0.991 between diversity index and evenness index (significant at the 0.01 level).

On different terrains soil diversity index followed the sequence of plain > medium-gradient mountain > medium-gradient hill > valley floor > medium-gradient escarpment zone > depression > high-gradient mountain (Table 9.2). The diversity index and evenness index had the same variation trend. The correlation coefficient was 0.839 between diversity index and evenness index (significant at the 0.01 level), similar to the above results from different parent materials.

3.1.3 Evenness

The evenness index of ultrabasic igneous rock (0.91) was the highest for all soil-parent materials, indicating that there was little difference between soils derived from this kind of rock. There were three groups of soils derived from ultrabasic igneous rock: Usti-Orthic Primosols, Hapli-Ustic Argosols and Calci-Ustic Argosols (Cooperative Research Group 2001). The area ratios of these soils are 3:3:1. The evenness index of soils derived from acidic igneous rock was 0.72. There are four soil groups derived from acidic igneous rock, including Usti-Orthic Primosols, Albi-Udic Argosols, Hapli-Ustic Argosols and Calci-Ustic Argosols with the area ratio of 11:6:1:1. The evenness indices of soils derived from intermediate igneous rock and unconsolidated material were low, 0.37 and 0.27, respectively, which indicated that areas occupied by different soils derived from these two kinds of rock or materials varied greatly. There were six groups of soils derived from unconsolidated material (Ochri-Aquic Cambosols, Hapli-Ustic Cambisols, Aquic Orthic Halosols, Usti-Sandic Primosols, Dark Aquic Cambosols and Hapli-Alkalic Halosols) with an area ratio of 652:65:19:6:2:1. All their sub-groups area ratios supported their evenness indices well.

The evenness index of soil developed on medium-gradient mountain areas was the highest in the three kinds of terrains and were as high as 0.84. Four groups of soils (Hapli-Ustic Argosols, Albi-Udic Argosols, Ferri-Udic Argosols and Usti-Orthic Primosols) with an area ratio of 5:5:2:1, showed a well-balanced distribution. The evenness index of valley floor was 0.74 and the area ratio 15:1:1. The evenness index of high-gradient mountain areas tended toward 0; only one soil group (Usti-Orthic Primosols) occur on this kind of terrain.

3.1.4 Abundance distribution

In ecological studies, abundance distribution of individual species in a given community is an important issue. Ecologists have used four abundance distributions: geometric series, logarithmic series, logarithmic normal distributions and broken stick model (Sugihara 1980). The geometric series is the least equitable, that is, a few objects are dominant whilst the rest are very rare or infrequent. In the logarithmic series and the logarithmic normal distribution, objects with intermediate abundance are most common. The most equitable distribution is the broken stick model. All object abundance distributions mentioned above have their own characteristic curves. Object abundance distribution is usually analyzed on a rank/abundance plot.

The abundance of each soil group was plotted on a logarithmic scale against the decreasing order of the soil group's rank in abundance. Object abundance was expressed in percentages so as to provide a more direct comparison between soil landscapes with different areas. The abundance distributions of soils derived from the four parent materials matched the logarithmic normal distribution. Similarly, the abundance distributions of soil groups developed on level lands and sloping lands and those on plain, medium-gradient escarpment zone and medium-gradient mountain fitted the logarithmic normal distribution. These data are in agreement with previous biodiversity and pedodiversity studies (Ibáñez 1995b, 2005a).

3.2 The result of rapid urbanization on regional pedodiversity patterns and related soil resource implications

Based on the dataset of thematic mapper (TM) images of four different periods (1984, 1995, 2000 and 2003) in Suzhou and three different periods (1984, 1995 and 2003) in Nanjing, the urban expansion of Suzhou and Nanjing in the past 20 yr were analyzed. The spatial-temporal dynamic changes of pedodiversity patterns are shown by the diversity measure methods. A grey correlative analysis model (Sun et al. 2006, Zhang et al. 2006, 2007, 2010) was used to measure the impact of urbanization on pedodiversity change. Soil composition and nestedness of 70 town-level units in Nanjing were analyzed (Wang et al. 2006, 2007a,b). In Nanjing there are 47 pedotaxa types grouped into four classes, including superior, dominant, rare, and endangered.

Pedological assemblages show the existence of an incomplete nested subset pattern (nestedness pattern analysis is explained by Ibáñez et al. (2005a,b and Ibáñez et al. in this book). Some soil taxa were selectively occupied by the fast growing urbanization; natural soils were transformed from agricultural use to urban use, including roads and buildings. The areas of many soil types have been reduced greatly, some of which showed more than a 50 percent reduction in the area by the urbanization process only in the past 20 yr. This means that some soils are already becoming endangered, and this has received attention from the local public.

Rapid urbanization has caused many social and environmental problems including a clear loss of certain soil types or unique soil units. In Nanjing two soil types, Clay loamy fomic-ferriudic argosol and Loamy car-mottlic-fomic-orthic anthrosol (Cooperative research group 2001), may be in danger of disappearing under urban/suburban structures because they have been decreased by 41.4 and 62.4 percent respectively in the past 20 yr (Zhang et al. 2007). The areas of seven soils have decreased by more than 10 percent, and eight others have decreased by more than 5 percent.

Land-use changes, especially those resulting from the rapid urbanization process, have often had a great impact on pedodiversity. The loss of soil types may, therefore, represent loss of whole biological communities unique to those soil types. The conservation of pedodiversity also brings into question the wisdom of converting into agriculture those soils that have not previously been cultivated. This chapter uses the nestedness analysis method from biological studies to examine the spatial-temporal change of pedological assemblages and pedodiversity characteristics (nested subset patterns) from the rapid urbanization of Nanjing in the past 20 yr.

The data used in the study are a set of Landsat satellite TM images overlaid with the digital soil database map (at scale 1:200,000) in which 19 soil mapping unit delineations forming 869 polygons excluding 16 non-soil polygons were linked with their attributed databases of natural conditions and different soil properties of the studied area Nanjing.

The nestedness analysis method has been used in biological studies for many decades and is thought to have greatly expanded ecologists' capacities to deal with complex biotic patterns within archipelagos or "islands" of terrestrial or aquatic habitats. Diverse biotic and abiotic processes are believed to generate nested distributions, including non-random extinction, differential colonization, and nestedness of critical resources. Biodiversity research is the keystone of conservation biology and natural reserves design. One of the most obvious causes of variation in the distribution of plant species and communities is the underlying soil variation. Usually ecologists use soil variables, but do not consider soil types in the same way as they consider bio-species. Only recently, nested subset analysis has been applied to pedodiversity studies (Ibáñez et al. 2005b).

The methodology of nested subset analysis (see Ibáñez et al. chapter in this book) consists of analyzing data with a geographic information system (GIS) to combine urban land-use maps derived from satellite images from different times with data on soil characteristics contained in the established soil databases. The integration of satellite remote sensing and GIS technology has proved to be an efficient method for mapping and analyzing urban land-use change. Some ideas from SOTER methodology were borrowed to build a database for spatial analysis and evaluation. In a perfectly nested matrix, the hypothetical line that separates the occupied area of the matrix from the unoccupied portion is called "boundary line" (see Ibáñez et al. chapter in this book). Taxa absences above and to the left of the line are defined as unexpected; the same is true for taxa presented below and to the right of the line. When randomness is low, unexpected presences and absences cluster near the boundary line. In contrast, when it increases, both unexpected presences and absences move further from it. The "entropy" of the matrix is a result (Ibáñez et al. 2005b).

The urban sprawl data from 1984 to 2003 was obtained by interpretation of Landsat TM satellite images from different dates (1984, 1995, 2000 and 2003). The images show Nanjing undergoing rapid urbanization in the past 20 yr. The city urban area increased 16,776 ha from its starting area 31,634 ha in 1984 to 48,410 ha in 1995 with a 1525 ha yearly increase (4.8 percent). During the next two periods, 1995–2000 and 2000–2003, all values were clearly higher than the previous annual increase: 2470 ha y^{-1} or 5.1 percent during 1995–2000 and an annual increase of 4804 ha y^{-1} or 7.9 percent during 2000–2003. In the past 20 yr, there has been a huge urban area increase of 43,544 ha in the Nanjing area, which is now twice as larger than before. More urban areas, including the city and attached county seat and township areas, expanded at an annual rate of 6.9 percent. To further examine trends in urbanization of the city, the data show that the city is more urbanized southward than in other directions because the city has been stopped in its growth by the Yangtze River to the northwest. It is considered to be the geographic line dividing North and South China and the mountainous area east of the city. People in this area traditionally preferred to live or do business on the south side of the Yangtze River, which played an important role in shaping the city structure in its recent history. The newly established Jiangning economic development zone (Jiangning District) south of the original Nanjing city is another reason for the current city urban/suburban growth pattern. Satellite-based calculations show that 11.3 percent of the total land area of Nanjing city is now in urban use (2003) compared with only 4.8 percent in 1984. The best soils are being impacted first even though the percentage of the urbanized area relative to the city total land surface is not very high. There is evidence, however, that some preservation of the better soils has been taking place.

Ecologists and pedologists are both aware that not all taxa that occur are widespread within the region. For a set of sites, one can envision a presence-absence, or incidence matrix, of resident taxa. Rows represent the sites, and columns represent names of all the taxa. Each site-taxon combination is represented by a one or a zero, depending on whether the taxon is present or absent at that site. Summing across rows gives the taxa richness recorded for a given site. Summing across columns for a given taxon gives the number of locations where its presence has been recorded. These matrices provide a simple graphic illustration of the interrelations between patterns of taxa and occupancy. The site sample describes the assemblages (pedologic or biologic) of the landscape in a probabilistic manner.

In this case study, all the calculated town level units are defined as “urbanized” or “not urbanized” (with zeros). Analysis of the expansion of Nanjing in the last 20 yr, results (Table 9.3) shows that the fill of urbanization analyzed by the Nested Temperature Calculator software (Atmar and Patterson 1995) has doubled from 3.5 in 1984 to 7.2 in 2003; and the nested degree has been growing with time, being the temperature (T) and indicator of randomness or disorder (T value at 18.68, 16.96, 16.40 and 15.17 in the four periods). T values range from 0°C in perfected nested matrices to 100°C in purely random matrices. The pattern of the land-use is all nested in 1984, 1995, 2000, 2003, and the nested degree has increased rapidly; Geographical changes can be found by the distribution of the fill value of different towns. Some towns or districts show a very clear increase in fill due to the more rapidly growing urbanization. Jiangning DZ ranks the first since it is a completely new development zone only in the past 10 yr while the others are less changed.

Similar to the method design above, all of the calculated soil mapping units at the family level are defined as urbanized or as agricultural (Fig. 9.1). Of 32 soil families, only two were not occupied by urbanization in the past 20 yr, including Loamy mollic-car-udic-orthic primosol in hilly areas and clay loamy typic-dark-aqui cambosol along the river basins while all the other soils, most of which are man-made paddy soils found commonly in the Yangtze delta, are more or less impacted first by the urbanization process. Loamy typic-Fe-leachic-stagnic anthrosol, clay loamy fimic-ferri-udic argosol and clay loamy car-typic-hapli-stagnic anthrosol occupied the largest urban areas of 4900 ha, 4660 ha and 3010 ha, respectively. According to the composition of pedotaxa, we use four classification classes (dominant, normal, rare, and endangered) in Nanjing area; four of 32 soil families are classified as dominant ones (D), 16 as normal ones (N), 10 as rare ones (R), and 2 as endangered (E), based on their frequencies of present and remaining areas. The most frequently appearing soil families are Clay loamy typic-arp-udic argosol, Loamy typic-Fe-leachic-stagnic anthrosol and Clay

Table 9.3 Nestedness analysis of land-use pattern under urbanization from 1984 to 2003.

Year	Matrix results		System temperature (Monte Carlo simulation)		Statistical significance
	T value	Fill value	Average	S.D	
1984	18.68	3.5	14.81	0.53	<0.001
1995	16.96	5.4	24.20	0.66	<0.001
2000	16.14	6.1	27.35	0.67	<0.001
2003	15.17	7.2	32.62	0.73	<0.001

loamy eutric-arp-udic argosol at 56, 52 and 41 percent, respectively. The least frequently appeared soil families are Clay car-vertic-gleyic-stagnic anthrosol and Clay loamy typic-dark-aqui-cambosol at 2 and 3 percent. The urban sprawl has turned one soil family from dominant into normal, two from normal into rare, and one from rare into endangered. Running the Nested Calculator program for the data from 1984 to 2003 shows that data sets are nested. The patterns of the composition and distribution of soils are all nested in 1984, 1995, 2000, 2003, and the degree of nesting has grown steadily. The nested metric T of soil composition for these four years is 16.88, 13.91, 13.62, 12.88, respectively. Area size and geographical conditions are considered to be the main factors forming the nested pattern in Nanjing area. Likewise there is a positive correlation taxa-area for this data set in Nanjing area, being the best fit a power law:

$$S=0.1096A^{0.5023} \quad F=126.64, \quad P<0.001, \quad R^2=0.8162$$

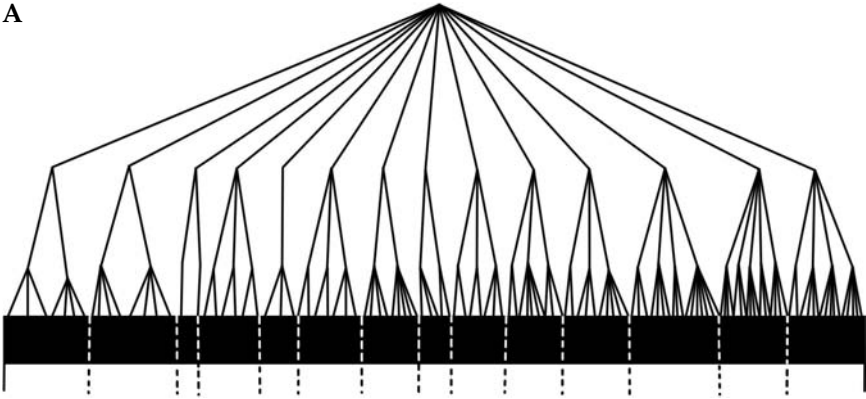
The spatial distribution of pedotaxa shows that the data set of the study area conforms to a hollow or Willis curve (see Ibáñez et al. Chapter 1 in this book), as is also the case of most biological and pedological inventories. Furthermore, this distribution conforms to a power law confirming what the ecological and pedodiversity literature predicts (Ibáñez et al. 2005b).

3.3 Comparison of diversity characteristics between pedological and biological taxonomies

Similar to the studies of Ibáñez et al. (2006a,b), a comparison of diversity characteristics between pedological classification system Chinese Soil Taxonomy and a biological taxonomy such as Tylenchina suborder of nematodes (Ibáñez et al. 2006a) was conducted (Zhang et al. 2008). The mathematical tools used are the same as in pedodiversity analysis.

Two comparative data sets from the Chinese Soil Taxonomy and biological taxonomy Tylenchina suborder are listed before the bifurcation ratios, Shannon index H' and evenness index E were calculated for both taxonomic constructs. Tables 9.4–9.10 and Fig. 9.2 show that the mathematical structures of both classifications are similar, user-friendly and efficient information systems. The main difference appears in the bifurcation ratios, with subtaxa per taxa of the Tylenchina suborder showing more profuse branching than in the Chinese Soil Taxonomy. The same was true for this suborder of nematodes in comparison with the USDA Soil Taxonomy (Ibáñez et al. 2006a).

A



B

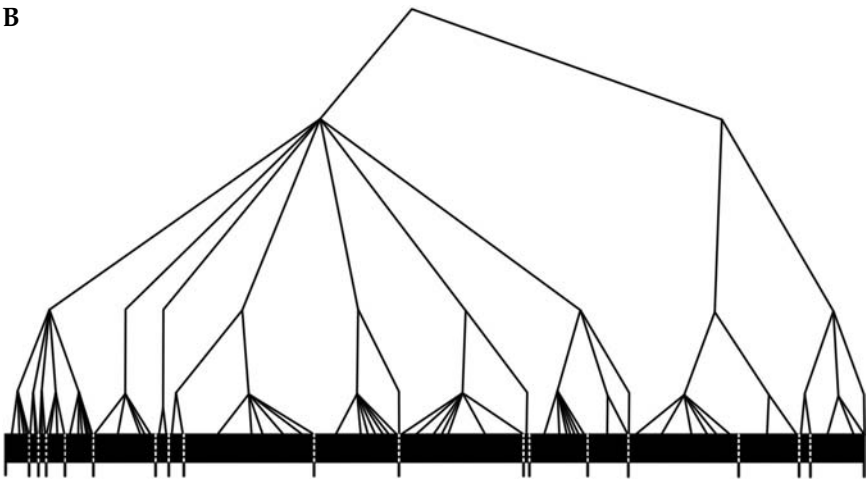


Figure 9.2 Branching system of the Chinese Soil Taxonomy (A) and branching system of the Tylenchina suborder of nematodes (B).

4. Future Challenges

China has been experiencing rapid changes in the past 30 yr since opening up to the outside world, leading to a real land-use change that is also very different between developed and less developed areas. This has caused considerable impact on pedodiversity patterns, which is now receiving more attention from scientific and governmental agencies. Under the new NSFC program, another meaningful new step has just begun to conduct the

Table 9.4 Number of taxa in each hierarchical category of the Chinese Soil Taxonomy and of the Tylenchina suborder of nematodes.

Category	Numbers
Chinese Soil Taxonomy	
subgroup	588
group	138
suborder	39
order	14
Tylenchina suborder of nematodes	
species	1955
genum	121
subfamily	19
family	9

Table 9.5 Bifurcation ratios (*BR*) for the Chinese Soil Taxonomy.

Data set	N1/N2	N2/N3	N3/N4	N4/N5
The whole soil taxonomy	4.26	3.54	2.79	14.00
Histosols	4.57	3.50	2.00	
Anthrosols	5.00	4.00	2.00	
Spodosols	1.50	1.00	2.00	
Andosols	3.17	2.00	3.00	
Ferralsols	5.33	3.00	1.00	
Vertosols	2.86	2.33	3.00	
Aridosols	5.11	4.50	2.00	
Halosols	3.60	2.50	3.00	
Gleyosols	3.57	2.33	3.00	
Isohumosols	3.50	3.33	3.00	
Ferrosols	6.20	3.33	3.00	
Argosols	4.18	4.25	4.00	
Cambosols	4.57	5.60	5.00	
Primosols	3.84	4.75	4.00	

$BR = N_i / N_{i+1}$; N1: Subgroups; N2: Great groups; N3: Suborders; N4: Orders; N5: The whole soil taxonomy

correlative analysis of pedodiversity and land-use type spatial distribution diversity at different scaled case areas in central and east China.

Yabuki et al. (2009) published a report online about pedodiversity and land-use diversity based on the data of 10 cities in Hokkaido, Japan and proposed using the concept of mutual entropy to evaluate the connections

Table 9.6 Bifurcation ratios (*BR*) for the Tylenchina suborder of nematodes.

Data set	N1/N2	N2/N3	N3/N4	N4/N5	N5/N6
Tylenchina suborder	16.16	6.37	2.11	4.50	2.00
Tylenchoidea superfamily	14.07	7.21	2.00	7.00	
Criconematoidea superfamily	26.70	4.00	2.50	2.00	

$BR = N_i / N_{i+1}$; N1: Species; N2: Genus; N3: Subfamily; N4: Family; N5: Superfamily; N6: The whole Tylenchina suborder

Table 9.7 The maximum entropy (H_{max}), Shannon index (H'), and evenness (E) of the Chinese soil taxonomy. The analysis refers to the figures of subgroups per great group, great groups per suborder, and suborders per order.

Data set	H_{max}	H'	E
subgroup/group			
The whole soil taxonomy	4.93	4.52	0.92
Histosols	1.95	1.87	0.96
Anthrosols	2.08	2.03	0.98
Spodosols	0.69	0.64	0.92
Andosols	1.79	1.72	0.96
Ferralsols	1.10	1.04	0.95
Vertosols	1.95	1.90	0.98
Aridosols	2.20	2.08	0.95
Halosols	1.61	1.43	0.89
Gleyosols	1.95	1.87	0.96
Isohumosols	2.30	2.22	0.96
Ferrosols	2.30	2.17	0.94
Argosols	2.83	2.77	0.98
Cambosols	3.33	3.27	0.98
Primosols	2.94	2.90	0.98
group/suborder			
	3.66	3.56	0.97
suborder/order			
	2.64	2.57	0.97

between the soil and land-use. Duan and Zhang (2011) modified this idea to analyze pedodiversity and land-use diversity of Nanjing in 1988 and 2003 before exploring the relation between soil and land-use diversity patterns. There will hopefully be some new findings in the near future.

Table 9.8 Maximum entropy (H_{max}), Shannon index (H'), and evenness (E) of the Tylenchina suborder of nematodes. The analysis refers to the figures of species per genus, superfamily, and subfamilies per family.

Data set	H_{max}	H'	E
species/genus			
Tylenchina suborder	4.80	3.76	0.78
Tylenchidae family	3.37	2.75	0.82
Anguinidae family	2.30	1.03	0.45
Beonomidae family	1.10	0.92	0.84
Pratylenchidae family	2.64	1.72	0.65
Hoplolaimidae family	2.08	1.43	0.69
Heteroderidae family	2.40	1.28	0.54
Criconeematidae family	2.40	1.84	0.77
Tylenchulidae family	2.20	1.34	0.61
superfamily			
Tylenchoidea superfamily	4.57	3.48	0.76
Criconematoidea superfamily	3.00	2.29	0.76
subfamilies/per family			
Tylenchina suborder	2.20	2.08	0.95

Table 9.9 Correlation coefficients of the fits to the power law and lognormal distribution of the Chinese soil taxonomy and the K and D power law coefficients. The analysis refers to the figures of subgroups per great groups.

Data set	Power law	K	D	Lognormal
The whole soil taxonomy	0.90	42.63	0.36	No fit
Histosols	0.91	14.70	0.40	0.94
Anthrosols	0.97	15.93	0.24	0.74
Spodosols	1.00	2.88	0.33	0.41
Andosols	0.92	9.56	0.52	0.89
Ferralsols	0.89	10.37	0.69	0.67
Vertosols	0.87	12.88	0.29	0.71
Aridosols	0.95	19.52	0.32	0.76
Halosols	0.85	18.60	0.43	0.78
Gleyosols	0.84	10.63	0.37	0.56
Isohumosols	0.87	14.74	0.25	0.81
Ferrosols	0.82	26.20	0.33	0.59
Argosols	0.98	22.04	0.56	0.98
Cambosols	0.98	39.56	0.27	0.98
Primosols	0.97	23.48	0.21	0.97

K = empirical constants, D being the regression line exponent or fractal dimension

Table 9.10 Correlation coefficients of the fits to the power law and lognormal distribution of the Tylenchina suborder of nematodes and the K and D power law coefficients. The analysis refers to the figures of species per genus.

Data set	Power law	K	D	Lognormal
Tylenchina suborder	0.93	221	0.72	No fit
Tylenchoidea superfamily	0.94	210.1	0.80	0.93
Tylenchidae family	0.93	35	0.79	0.91
Anguinidae family	0.99	102.3	2.20	0.97
Beonomidae family	0.97	16.27	1.14	1.00
Pratylenchidae family	0.98	156.63	1.41	0.98
Hoplolaimidae family	0.98	96.8	1.40	0.98
Heteroderidae family	0.99	174.71	1.65	0.99
Criconematoidea superfamily	0.95	139.8	0.88	No fit
Criconematidae family	0.95	135	0.99	0.94
Tylenchulidae family	0.97	77.83	1.44	No fit

K = empirical constants, D being the regression line exponent or fractal dimension

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Conclusions

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This book analyzes the state of the art in pedodiversity studies and its multiple applications to geology, ecology and other disciplines as it is showed in several of its chapters. Chapter 1, written by Ibáñez and co-workers, introduces this topic.

In the first instance the concept of diversity seems clear and intuitive. However, this is not the case. Many definitions have been proposed in the literature but all of them have been a subject of controversy between rival schools. In fact, when research has problems to carry on the procedures for operationalization of a concept implies that many proposals are done by rival schools often positioned in irreconcilable approaches (see Chapter 1).

The quantification of diversity suffers from the same problems as its definition. Because the concept of diversity suffers from unavoidable value judgments, hundreds of indices, distribution-abundance models and other mathematical formulations have been proposed in the past few decades, but none of these appear to be superior to the rest. Enrico Feoli and co-workers carry out a deep innovative analysis of this topic in Chapter 2.

Because humankind is changing and has degraded the Earth surface, researchers, policy makers and the public are concerned with the deterioration and loss of natural resources. In this framework, pedodiversity analysis is a relevant indicator of the environmental health at local, national, regional and global scales. Biodiversity has gained attention over the past six decades, whereas soil diversity only received the attention of a small group of pedologists in the last 20 yr. For this reason, pedodiversity is in its infancy with respect to biological diversity studies.

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The preservation of the pedosphere and its diversity is of great importance for the survival of the humanity and for sustainable development. Pedodiversity is part of our natural and cultural heritages. Pedodiversity is also an essential part of our biological heritage in view that soils are more biodiverse than aboveground plants and animals and constitute the metabolic apparatus of ecosystems. Likewise many aboveground species are considered soils' endemism and thus if the soil is degraded or lost the latter disappear (see Chapters 1, 5 and 8).

Lo Papa and Dazzi as well as Xuelei in their respective chapters show that there are many natural pedotaxa at risk of extinction as a result of intensive industrial farming practices, as well as urban sprawl and the development of human infrastructures. Regrettably the lack of detailed and updated inventories do not permit the quantification of this process. In any case the above mentioned authors detect an initial increase of pedodiversity by the creation of manmade soils, but this is followed by a progressive decrease of pedodiversity. Furthermore using mathematical forecasting tools, Lo Papa and Dazzi predict the loss of pedodiversity in the coming decades. In addition Chapter 8, written by Bockheim and Haus, explains the concept of soil endemism, the importance of preserving rare and endangered soils, and the approach for quantifying their patterns.

Pedodiversity is also part of our geological heritage. In the early 1990s, the term "geodiversity" was introduced by geoscientists who realized the analogy between biotic and abiotic components of the natural environment (e.g., Sharples 1993). As also occurs in biodiversity analysis, many definitions of geodiversity have been proposed (e.g., Gray 2004). The early definitions consider geology, landforms and soils as the main components of geodiversity. However, in the last few years, a more holistic approach has been used, which incorporates many other resources and processes, such as topography, hydrology, climate and biological fossils (Serrano-Cañadas and Ruiz-Flaño 2007). As a result of a plethora of proposals, the quantification of geodiversity has become a very difficult task. Some experts have proposed developing complex indices to include all these structures and processes in a single value, comparing this type of "abiotic diversity or heterogeneity" with biodiversity ones (Gray 2004). Such an approach regarded the geodiversity framework as a major challenge to introduce the abiotic nature in the main stream of ecological literature (Petrisor and Sâsrbu 2010). Four aspects are common to most geodiversity analyses: (i) a parallelism between biodiversity and geodiversity; (ii) a strong conservation focus on preservation of our geological heritage; (iii) a consideration of richness, but in general a lack of attention to the abundance distribution of objects among categories or taxa; (iv) examination of soil attributes, but

not soil taxa. In our opinion, the main shortcomings of these approaches are: (i) they include a variety of natural resources that vary in discrete and/or continual natural ways; (ii) the lack of universal classifications of many geological resources hinders comparisons among different studies; (iii) there is no agreement on what abiotic resources should be considered; (iv) numerical values generated by the proposed indices are difficult to interpret. Regrettably the mainstream of geodiversity literature has not paid attention to soil diversity analysis, despite its definition, and is a less mature line of research from the point of view of its mathematical formalization.

It is clear that many soils and paleosols are a block of memory or an archive of past environments, ecosystems, and climates; thus if the former are lost or degraded, these natural archives are lost as a part of our natural history register (see Chapters 1, 5, 8).

However pedodiversity is also a part of our cultural heritage. Nowadays studies show that many ancient farming practices were sustainable for centuries or millennia. Furthermore ethnoagriculture and ethnopedology show that aborigine cultures across several continents built effective artificial manmade soils (including some which conform to the criteria to be considered Technosols by the WRB). On the other hand, many industrial agricultural practices are degrading and contaminating soils and thus could be considered unsuitable for a genuine sustainable development. Thus an inventory or a "red book" of ancient agricultural sustainable practices and its "artificial" soils seems currently imperative (Ibáñez et al. 2008). Some studies show that several native soil classifications are equal to or better than the modern ones. Thus, for example, some indigenous classifications recognize more soil classes and fragments of the local soil cover in a more detailed way than modern international classifications (Bautista and Zinck 2010). Diagnostic properties used to recognize, distinguish and classify soils are relatively similar for both the indigenous and scientific approaches. It would be interesting to know what pedodiversity patterns could be detected using some of indigenous classification systems. Regrettably this is a hole that has been not filled to date in pedodiversity analysis. Furthermore, the number of pedotaxa of this guise remains to be identified, inventoried and classified.

Pedodiversity inventories can also be used to design networks of soil reserves that conserve soil types as natural (non-disturbed) as possible. This initiative could be useful to (i) preserve soil diversity as well as the living organisms that depend of a healthy soil system for survival and (ii) identify benchmark soils for soil monitoring programs, comparing natural versus domesticated pedotaxa. In Chapter 5, Saldaña carries out a comprehensive analysis of the design of networks of soil reserves as well as the role of pedodiversity analysis from the landscape ecology perspective.

In Chapter 2 Feoli and co-workers present a synthesis of diversity tools that are presently available, as well as the explicit and/or underlying relations among them. This list is expanded in other chapters as in the case of Ibáñez and co-workers and Saldaña, among others. In general all diversity could be analyzed using the same mathematical formulations. This fact permits the comparison of diversity patterns detected in different natural resources such as biodiversity, pedodiversity, landform diversity and lithodiversity. As a general rule it seems that biodiversity and pedodiversity follow the same mathematical patterns as is shown in the chapter by Ibáñez et al. Furthermore, Toomanian detects the same similarities with his first test of landform diversity. In addition there is evidence that litho-diversity could also follow the mathematical regularities detected in biodiversity and pedodiversity studies. Landforms and lithological diversities deserve more attention from experts in their respective disciplines. This type of analysis could lead to some intriguing areas deserving further research. The causes of these mathematical regularities could suggest that the structure and dynamics of Earth surface systems obey laws that go beyond those currently identified in specific disciplines and what might be precursors of other more general ones, such as in physics of non-linear systems and complexity science. In the latter case some regularities could be idiosyncratic at some scales whereas others are scale-invariant (i.e., fractals and multifractal structures). In his chapter Phillips explains the nonlinearity of soil-regolith and landforms diversities, whereas San-José and Caniego show the fractal and/or multifractal nature of biodiversity and pedodiversity patterns, among others that form part of the Earth surface systems. Therefore we are involved in the exciting possibility of finding and applying a general theory of diversity that covers many natural resources simultaneously. However, in order to corroborate or refute this conjecture proposed some time ago by Ibáñez et al. (1990, 1998) and Phillips (1999) it will be necessary to conduct more studies in the coming decades.

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Color Plate Section

Chapter 8

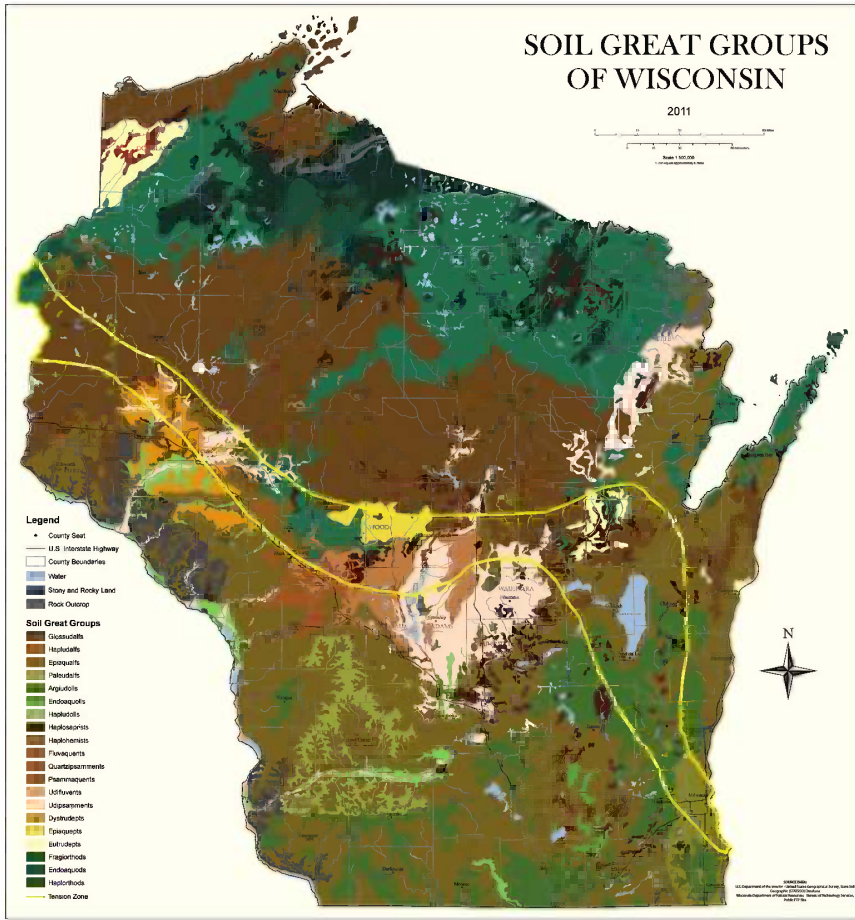


Figure 8.3 Soil great-group map of Wisconsin showing the ecological tension zone.

The preservation of natural resources and their respective diversities is a vital component of sustainable development. Soil diversity, or pedodiversity, is part of our natural (geological and biological) and cultural heritage (manmade sustainable soils). The preservation of the pedosphere is essential for the biosphere and Earth's systems, the regulation of the climate system, as well as for the world food security. Likewise pedodiversity studies, as is shown in this book, are invaluable tools in pedology, spatial soil pattern analysis and quantitative soil geography. Within this book many of the most reputed international experts on this topic discuss the state of the art of pedodiversity analysis. It is of interest that pedodiversity patterns conform to the same mathematical regularities detected in biodiversity patterns, challenging many current theories. The authors also analyze the relations among biodiversity, pedodiversity, landform diversity, lithodiversity, and land use diversity. The chapters of this book should be not considered as individual contributions but as a story told by its protagonists from their various perspectives along the same storyline. Thus this book, the first of its kind, is intended to be a combined handbook, historical account of pedodiversity research, and essay on its future challenges.

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