Chapter 9
Bioindication-Based Approaches for Sustainable Management of Urban Ecosystems

Andrey Gorovtsov, Vishnu D. Rajput, Sergey Gorbov, and Nikita Vasilchenko

Abstract Urbanized areas are covering less than 3% of the land, but the majority of Earth's population and industry is concentrated at these territories. There is an urgent need for development of a comprehensive approach to the assessment of environmental quality in these areas. Bioindication allows estimating the entire complex of negative factors simultaneously. However, there are still large gaps in our knowledge of the urban ecosystem functioning. This chapter aimed to review the existing approaches to the bioindication of urban areas, i.e., microbial and plant bioindicators, as well as complexity of urban ecosystem, soil and its types, anthropogenic impacts, pollutants, effect on microbial community, other existing problems in this field and suggest the possible ways to solve them. The development of reliable bioindicators used on the basis of systematic approach would contribute greatly to rational land use and sustainable functioning of the urban environment.

1 Introduction

Although the cities cover a very small share of the world territory, they are home to great numbers of people. According to modern estimations, more than 60% of world population will inhabit urbanized areas by the year 2030 (Alberti et al., 2003). In industrialized countries, the percent of urban population is already very high. By 2015 over 90% of Belgium, Uruguay, Argentina, Brazil, Japan, South Korea, and Israel population inhabited cities and suburban areas. Thus, the importance of studies dedicated to urban ecology and sustainability of urban ecosystems is rapidly growing. Our knowledge of the urban ecosystems and the laws of their functioning and development remains insufficient. This leads to difficulties in urban planning and low sustainability of the urban environment.

A. Gorovtsov (✉) • V.D. Rajput • S. Gorbov • N. Vasilchenko
Academy of Biology and Biotechnology, Southern Federal University,
Rostov-on-Don, Russia
e-mail: gorovtsov@gmail.com

R. Singh, S. Kumar (eds.), *Green Technologies and Environmental Sustainability*, DOI 10.1007/978-3-319-50654-8_9

The starting point of any development is the assessment of the existing state of the system. In a complex and highly heterogeneous environment of a city too many variables have to be measured, and it is hardly possible to estimate the results of the interaction of many different factors. Bioindication allows an approach to assess the entire complex of negative factors simultaneously. Moreover, the use of living organisms for the estimation of the environmental quality can make it easier to predict the possible negative effects for the inhabitants of the city.

2 Urban Ecosystems

It can be stated that urban ecosystems have become the main habitat for humans. The rapid urbanization and migration from rural areas to cities can be recognized as the most significant human ecological event of the past 100 years (Rees 1997). This process, along with industrialization and technical development has led to formation of a principally new type of environment, with specific traits in biogeochemistry, energy consumption and even climate. Presumably, it will also have a strong impact on evolution of many species in urbanized areas, first of all on humans, and the long-term consequences can hardly be determined.

There is a set of principal factors that determine the formation and evolution of any ecosystem (Chapin et al., 2011). These are:

1. The climate of the territory, where the ecosystem is located
2. The substrate of ecosystem (soils and sediments)
3. Biota
4. Relief
5. The time (age of ecosystem from its formation, succession phase, etc.)

Urban ecosystems are affected by the same set of factors, which are transformed due to anthropogenic impact. The most prominent trait of urban climate is the formation of heat islands. The temperature in most cities is usually 1–2 °C higher than in surrounding rural areas.

The heat island phenomenon is due to several factors, including the lower vegetation cover and darker surface materials in the urban landscape (Akbari et al. 2001). Urban surface and air temperatures also increase due to soil sealing and decreases in evapotranspiration (Alcoforado and Andrade 2008). Urban areas also usually have decreased albedo, which contributes largely to the temperature rise. The differences in climate between city and countryside have biological and human implications (Pickett et al. 2011). Regarding the plants, increased temperature leads to later leaf drop time, earlier leaf emergence and flowering in the regions with moderate climate. Some migratory bird species can stay in the cities instead of using traditional migration roots and feed on landfill sites (Gilbert et al. 2016). There is also a distinct impact on soil. The increase of soil temperature alters soil microbial activity leading to enhanced nitrogen mineralization (Pouyat et al. 2003).

Large cities can also alter the atmospheric precipitation though it is largely dependent from world region (Mishra et al. 2015). It has been shown that urban heat island effects influence the rainfall largely during the early urbanization stage. However, at the stage of agglomeration formation, the regional moisture depression induced by the soil sealing has an effect on atmospheric instability energy, which might negate the city's positive impact on regional rainfall (Wang et al. 2015).

The hydrology of cities is completely different from rural areas. The main cause is the sealing of soils with asphalt, concrete, and other impermeable materials. Sealing of soil leads to drastically decreased evapotranspiration, while increasing the surface runoff. There are also major changes in groundwater flow, due to basement construction and reduced income from rainfall.

The soil is a very important part of urban ecosystem as it is the source of nutrients for urban vegetation, the basis for biogeochemical cycling and also the medium that accumulates and transforms various contaminants. Urban soils are considered as a separate soil type in many modern soil classifications (Prokof'eva et al. 2014), because of their deep transformation in urban environment. Taking into account the importance of soil for urban ecosystem sustainability and their convenience for bioindication will be discussed below in more detail.

In the urban ecosystems studies two approaches are usually considered. One defines the city as one single ecosystem; the other sees it as a composition of more or less separate ecosystems, such as parks, ponds, streets, city gardens, sewerage system, etc. (Bolund and Hunhammar 1999). We consider both approaches applicable, depending on the scale of study. When studying the so-called urban metabolism, including the inflow of resources, energy consumption and waste management, the city should be considered as an integrated heterogeneous system. Oppositely, most of the biodiversity studies face too much heterogeneity, when dealing with a whole city, because the type of land use differs sharply in different parts of the city, and it has a strong impact on animal, plant and microbial communities. This heterogeneity has led to attempts of classification of different parts of urban territories into several types, linked to descriptive studies of flora and fauna. For example, Dorney (1977) has classified urban landscapes into six groups creating a gradient from urban to rural ecosystems – business central part, old quarters, new quarters, construction sites, suburban areas, and rural areas.

The biodiversity of plants in the urban areas is quite high, and that species richness is mostly artificial—floral complexes of cities include large percentages of exotic species that were intentionally introduced. At the same time, the number of native species tends to decrease (Rapoport 1993). Thus, the urban ecosystems tend to harbor new plant biocenosis that are not characteristic of the surrounding rural area.

The biodiversity of most animal taxa is decreased in urban areas due to the loss of strictly specialized species. However, some species adapt to the conditions changed by urbanization process and colonize the available habitats rapidly. Therefore, diversity depends on the balance between extinction and colonization, which differ regionally and taxonomically. At moderate levels of urbanization, species richness may actually be higher than in nearby wild lands (Pickett et al. 2011).

We can conclude that urban ecosystems comprise a set of biotic and abiotic conditions that are substantially different from those occurring in natural environment. These systems are subject to the general environmental laws, but apparently have their own unique characteristics that require attention and investigation.

3 Urban Soils: The Basis of Ecosystem Infrastructure

Soils emerging in the urban environment act as a base component of the ecosystem. They are the central part of biogeochemical cycles, mediate the biochemical conversion of the cultural layer, the transformation of surface water into the ground water. They also act as a nutritious substrate for plants. The soil is a bank of seeds, the regulator of the gas exchange, etc. (Dobrovolsky and Nikitin 1990). Nevertheless, urban soils have a number of specific features, which leads to classifying them as a new type of soil.

Urban soils research began a long time ago, in the 60s of twentieth century (Zemlyanitsky 1963; Zelikow and Popkov 1962). However, at that time they were considered as a special case of natural soils that have been subjected to the influence of the urban environment. For the first time the definition of a new type of soil was proposed by Bockheim in 1974. According to this definition, the urban soil is a soil material with an artificial surface layer which does not possess agricultural value, which thickness exceeds 50 cm, formed by mixing, filling or pollution of the soil surface in the city or surrounding areas (Bockheim 1974). Sometimes it is also stated that a sign of urban soils, in addition to the above, is the presence of synthetic or toxic substances in quantities greater than those in the natural soils (Craul 1985a, b; Blume 1986; Burghardt 1996).

There is no common classification of urban soils at the moment. One of the reasons for this is the lack of a common approach to nomenclature and taxonomy of urban soils.

The most widely accepted system is "World reference base for soil resources"—WRB, adopted at the 1992 Congress in Montpellier in France. During the period 1998–2006, WRB acquired the status of a formal system of nomenclature and classification of soils in the European countries and the Central African Soil Science Association.

The main diagnostic horizon of urban soils is the horizon termed in Russian soil classification as "urbic" (UR)—sinlithogenic diagnostic horizon: gradually formed by bringing a variety of substrates to the surface in urban and rural settlements. It has often brownish-gray color tone, expressed on a scale of Munsell as follows: value (lightness) of less than 6, chroma (color) 1–4. The horizon contains more than 10% of artifacts (mainly construction and household waste), often sandy and/or rocky. The chemical properties are highly variable and evaluated in relation to the natural counterparts; usually the soil has a neutral to alkaline reaction, often calcareous. The content of individual chemical pollutants does not exceed 2 MPC. The content of available phosphorus (extractants 0.2 M HCl, 1% $(NH_4)_2CO_3$, 0.5 M $NaHCO_3$) is increased (on average no more than 0.1–0.2% (100–200 mg/kg). The humus content is highly variable, and its composition reflects zonal conditions (Prokof'eva et al. 2014).

The WRB classification system defines most urban soils as Technosols, while anthropogenically transformed agricultural soils fall into Anthrosols group. Technosols should contain artifacts, can contain pieces of rocks, and often contain toxic substances. Technosols include soils from wastes such as landfills or mine spoils, pavements and underlying materials, soils with geomembranes and constructed soils in human-made materials (Nachtergaele 2005 in Anne Naeth et al. 2012).

Urban factors directly and indirectly affect soil chemical, physical, and biological characteristics (Pickett et al. 2011). The direct impact on soil includes physical disturbances, i.e., mixing of the upper soil layer with various natural and artificial materials, burial and sealing of soils, creation of artificially constructed soil layers. The changes in the environment have various side effects on soils, which can affect pedogenic process and have long-term consequences. These changes include heat island effects, changes in moisture level, higher levels of nitrates in atmospheric precipitation, changes in biotic environment including new species of plants and animals.

4 The Sources and Types of Anthropogenic Impact on the Soils in Urban Areas

Same to the natural ecosystems, the urban ecosystems are shaped by a complex of environmental factors, but the anthropogenic influence becomes dominant in the city. While the water and the atmosphere are mobile parts of the environment, the soil is accumulating all the impacts, including those originating from atmospheric and water sources and can be used as a reliable object for the bioindication of the whole urban environment. Therefore, we focus on the impacts on soil. Anthropogenic impacts on the soil of a city can be divided into three types:

1. Chemical impact—the income of various pollutants linked to human activities into the soil.
2. Physical and mechanical—change of soil structure by direct mechanical action, exposure to sound and different types of electromagnetic radiation, including overheating.
3. Biological—the income of foreign soil microorganisms, reduction of organic matter income from leaf litter, introduction of exotic plant species and others. The following is a brief overview of these types of anthropogenic impacts on the soil in urban ecosystems.

4.1 Chemical Pollution in Urban Environment

4.1.1 Contamination of Soil with Heavy Metals

Since the sixties of twentieth century until now the environmentalists, urban planners, and soil scientists are interested in the problem of pollution of urban soils with heavy metals. It should be noted that this type of soil contamination is the most extensively studied, since almost every publication on urban soils contains information about the trace elements pollution. Many urban ecologists believe that all urban soils are contaminated with heavy metals.

Industrial emissions enter the soil with rainfall, the precipitating dust and aerosols, or by direct absorption of soil gaseous compounds. They can also be absorbed from the atmosphere by the plants, accumulate therein and be transmitted into the soil with the litter. Accumulating in the soil in large quantities, the heavy metals become quite mobile and can leach into the groundwater, causing the pollution to spread to remote areas from the primary source (Steinmann and Stille 1997; Wilcke et al. 1998).

Heavy metals involved in the biological cycle are transmitted by food chains and cause a number of negative consequences. At the maximum levels of chemical contamination the soil loses its ability to support the growth of plants and lacks biological self-cleaning. This can lead to a loss of ecological functions and death of the ecosystem. In addition, such heavy metals as Hg, Cd, Ni, Cr, Cu, Co, etc., cause carcinogenic, mutagenic, or teratogenic effects on humans. There is also evidence for neurotoxicity for some elements (Jooste et al. 2015). The key contaminating elements vary in different cities around the world, but most frequently the urban soils are polluted with lead, zinc and cadmium. Among the key heavy metal contaminants one can also mention Cu, Cr, Ni, Co, and Hg (Alloway 2013). The impact of different metals on the ecosystem is highly variable. For some metals, no distinct biological role is determined up to date. These are Sn, Ga, Zr, and members of lanthanoid element series. The contamination of urban environment with these metals is uncommon, and their toxicity is quite moderate. Fe, Mn, and Mo are important micronutrients with low toxicity level. However, many essential elements show high level of toxicity when their concentration is elevated. These are Zn, Cu, Ni, V, and Co. There is also a group of highly toxic elements with no significant biological role, i.e., Cd, Ag, Hg, Pb, Sb, and other elements (Wyszkowska et al. 2013).

4.1.2 Contamination of Soil with Non-metallic Elements

In addition to increased concentration of heavy metals, urban soils are also vulnerable to the adverse effects of a number of non-metallic elements. Especially dangerous is the pollution of urban soils with arsenic. Arsenic in urban environment is often preserved from the times when it was actively used as wood-protecting agent in construction materials. Also, arsenic compounds were used as pesticides and can be found in soils of old gardens, later included into the city borders (Elless et al. 2008). High content of arsenic was also observed near the metallurgical industries and enterprises, where the coal is burned (Lambert and Lane 2004). Arsenic contamination significantly affects the microbial activity and the composition of the microbial community. Its effects include sharp decline in urease, protease, and phosphatase activities in soil (Lorenz et al. 2006).

Another frequent problem is the chloride contamination. In many cities technical salt containing up to 99% of sodium chloride is used on the roads to combat the icing. Though chloride ions are not very toxic for humans, they can cause dangerous effects to the urban ecosystems. In the soil, increased concentration of chloride and sodium ions affects its structure, water and air permeability, osmotic potential and leads to the loss of stability of the soil, as well as osmotic stress for all the organisms inhabiting it (Černohlávková et al. 2008). Furthermore, the use of defrosting agents results in mobilization of heavy metals in the soil, improving their bioavailability and toxicity (Bäckström et al. 2004). In addition to sodium chloride, calcium chloride is often used as a de-icing salt. Excessive intake of calcium significantly increases the pH of urban soils, which affects the properties of the soil and its microbial community (Puskás and Farsang 2009). Salinization affects the proportions and species composition of microbial communities in the soil, while significantly decreasing their metabolic activity (Yuan et al. 2007).

4.1.3 Contamination of Soil with Organic Compounds

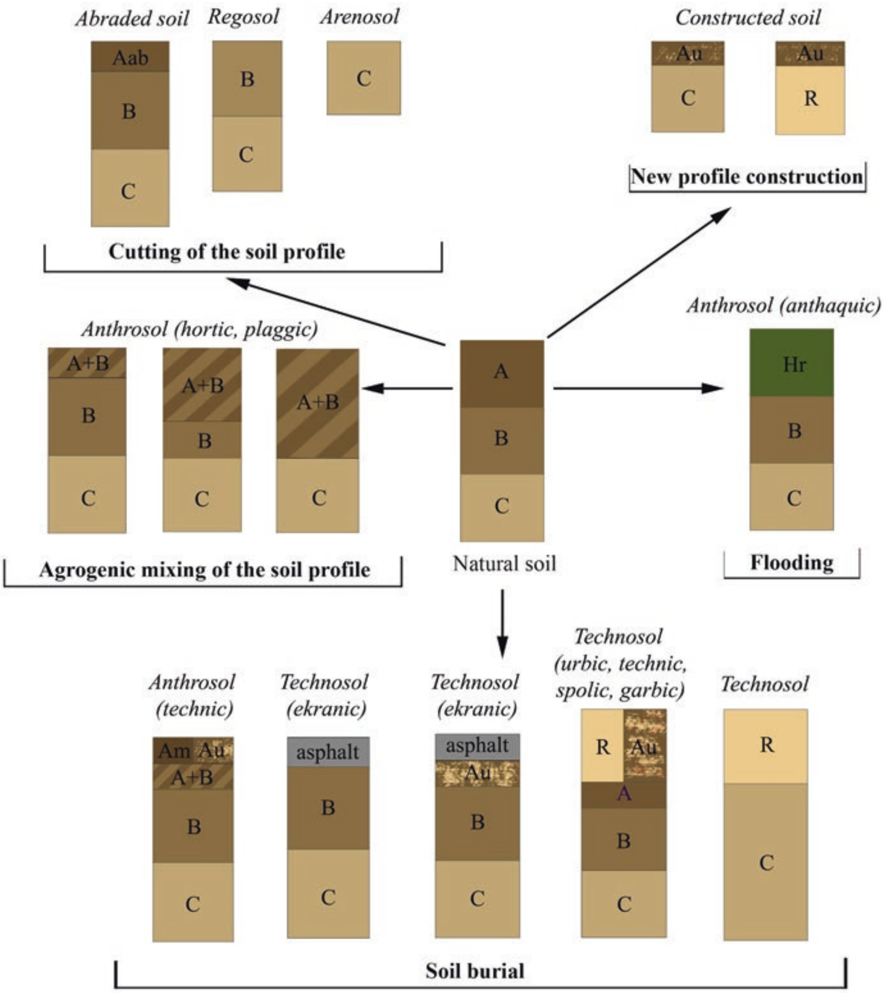
The global chemical pollution of the biosphere with synthetic organic compounds is one of the most acute problems of our time, causing justified concern about a possible violation of ecological processes and ecological balance in certain areas of the biosphere. With reference to urban soils, the following groups of organic contaminants are often considered in the literature: polychlorinated biphenyls, polyaromatic hydrocarbons, phthalates, dioxins and dibenzofurans, as well as aliphatic and alicyclic hydrocarbons, contained in oil products. The sources of polychlorinated biphenyls and naphthalenes in urban soils may be vehicles, burning of household waste, as well as industrial enterprises. In addition, these substances were for a long time included in the composition of technical greases, were used as wood-protecting agents and dielectrics (Krauss and Wilcke 2003). Polycyclic aromatic hydrocarbons are products of incomplete combustion of fuel in the engines of cars and industrial plants that use coal. These substances are almost exclusively anthropogenic accumulate particularly intense in urban soils. As well as polychlorinated biphenyls, polycyclic aromatic hydrocarbons have been shown to have mutagenic, carcinogenic and teratogenic effects, so their accumulation in urban soils is a serious threat (Aichner et al. 2007). The concentration of polycyclic aromatic hydrocarbons in urban soils often more than ten times exceeds their concentration in the natural soil. At the same time, the accumulation of them in urban soils with different land-use types also varies greatly. It reaches a maximum at the sides of the busy streets, significantly decreasing in residential and recreational areas. The decrease in the concentration of organic contaminants is also observed when moving from the center to the outskirts of cities (Liu et al. 2010). Another important class of organic pollutants are phthalic acid esters, which are widely used as plasticizers in a variety of types of plastics. Every year nearly six million tons of these materials are produced in the world. Phthalic acid esters have a hormone-like effect, and therefore can cause serious disruptions to the reproductive system of many organisms. Urban soils are considered as one of the main reservoirs of this type of pollutants, from where they can leach into the ground water or sublime to the atmosphere (Zeng et al. 2009).

4.2 Physical Disturbance of Soils in Urban Environment

4.2.1 Urban Pedogenic Process

The soils in urban environment can undergo radical transformation of profile under the influence of anthropogenic effects or direct construction of new soil profiles. Acquiring a new horizons system, which is unique in comparison to the natural soils leads to grouping such soils in special "man-made" departments with new names. Several types of restructuring of soil profile are possible (Fig. 9.1):

- As a result of deep and long homogenization of the upper part of the profile due to agricultural use of soils. At the same time the soil, depending on the thickness of the whole profile can lose the signs, allowing to identify its original typical identity. The mixed agrogenic horizon occurs on the remains of the median horizon or on soil-forming rock. Such soils in urban landscapes are identified as anthrosols (hortic, irragic, plaggic, anthaquic subtypes are determined based on the type of land use). These soils usually occur at the outskirts of the cities that had been involved into urbanization process in the past few decades.
- As a result of any mechanical abrasion or cutting the top natural horizons. On the surface there are median horizons, and the remaining soils are classified as regosols and arenosols.
- As a result of deliberate or provoked deposition of mineral (often containing humus) material on the natural or cutted surface of the soil. Such soils can be classified both as anthrosols and technosols, depending on the thickness and artifact content of the newly formed layer. Technosols usually contain 20% or more artifacts, while technic anthrosols contain less artifacts. The soil surface can be also sealed with impermeable layers (asphalt, concrete) forming an ecranic technosol profile.
- As a result of prolonged flooding, coupled with partial mechanical disturbance of the natural profile (usual suffixes "aquic" "anthraqic").
- As a result of creation of a new soil profile during remediation or construction of the urban environment. In some cases, the new soil profile is formed over an impermeable geomembrane to avoid the transport of hazardous substances from the underlying layers.



A - Topsoil
B - Subsoil
C - Parent material
R - Bedrock
A+B - New horizon created by agrogenic mixing oftopsoil and subsoil
Aab - Abraded topsoil
Au - New horizon created from topsoil due to anthropogenic activity, contains artefacts.
Am - Horizon created from topsoil due to high level of compaction
Hr - Horizon with strong reducing conditions created by long-term anthropogenic flooding

Fig. 9.1 Main types of soil profile formation in urban environment

4.2.2 Compaction of Soils

Most of the disturbing impacts on soil in urban areas are considered to be inherent to industrial areas or business center of the city with intense traffic. However, some types of anthropogenic impact are more significant in recreational areas, i.e., parks or city forests. The recreational use of these areas leads to compaction of soil surface layer because of intensive trampling. It has been shown that the level of soil compaction is much higher in neighborhoods with low socio-economical level, due to higher population density and lack of open spaces for walking and rest (Zhevelev and Bar 2016). There is also evidence for a positive correlation between soil pH level and soil compaction (Andres-Abellan et al. 2005). However, it is not clear whether this correlation is due to changes in the soil properties, or it is related to the positive correlation of higher pH values and the level and time of the urbanization process.

The level and the consequences of soil compaction are different in various microenvironments in the recreational areas. The paths and picnic areas are the most compacted, which leads to impairment of vegetation development and decrease in soil organic matter content (Sarah et al. 2016).

The negative effects of soil compaction include the decrease of soil arthropods biodiversity (Moriyama and Numata 2015), which leads to the further compaction of soil, because the soil animals play an important role in formation of soil structure. There is also some recent evidence for the loss of plant biodiversity in urban forests due to soil compaction (Vakhlamova et al. 2016).

4.2.3 Microclimate Changes Impact

One of the most important impacts on the urban ecosystem as a whole and particularly on urban soil is the local climate changes observed in urban areas. The most studied phenomenon is the urban heat island effect. This effect is mainly due to increased heat production by industry, lack of natural heat sinks, decrease of evapotranspiration-related cooling due to soil sealing, decrease in airflow because of dense building, increased solar radiation absorption by asphalt and roofs of the buildings, etc. The influence of increased temperature on urban environment has been at part discussed above in this chapter, and it is widely described in the literature.

The other microclimate parameters have been studied to a much less extent. The ventilation in the urban areas plays an important role for the nutrient cycling, pollution control and can have long-term effects on the whole urban ecosystem. There are specifically urban atmospheric phenomena, affecting the city environment, i.e., street canyons, which cause significant changes in the air circulation in urban areas. The airflow is usually significantly lower in urban areas than in the city surroundings. It has been shown that the wind speed in the city center can be as low as $\frac{1}{3}-\frac{1}{4}$ of the speed outside the city (Dimoudi et al. 2013). The decreased ventilation can lead to deposition of larger amounts of dust, which plays an important role in soil pollution and pH shifts in urban areas. The use of street vegetation strongly affects the rate of deposition and dispersion of dust in urban areas (Janhäll 2015). The vegetation filters the air, improving its quality, but at a cost of increased deposition of pollutants to the soil surface. It should be noted that the pollutants incorporated into the soil can be resuspended to the atmosphere with the soil-borne dust. A recent study has shown a correlation between lead concentration in soil, in the air dust, and in the blood of children, inhabiting the city, which was dependent from season (Zahran et al. 2013). Therefore, while the vegetation-based air filtration is important for the quality of the environment, the corresponding side effects should be taken into account. The proper design of the city environment can mitigate these effects. The urban planning should allow the clean air income, and the high trees should not block the airflow, especially in the streets with heavy traffic. This can improve the wind transportation of pollutants outside the city and decrease of their concentration below the harmful levels.

Some changes also occur in the hydrology of the urban ecosystem. These changes are highly dependent from the increased temperature, because of the growing transpiration rates which can lead to the depletion of the moisture in the surface soil layer by the grass (Arden et al. 2014). The other effects include the major changes in the water income due to the soil sealing. The large territories, covered with impermeable layers lead to uneven distribution of atmospheric precipitation, entering the soil. The areas that remain unsealed can be flooded by the rainwater from the adjacent sealed areas. This can lead to water erosion and the degradation of the topsoil layer, which is washed out to the roads. After drying, the washed soil can be turned into the dust and pollute the city atmosphere.

4.3 Biological Impacts in Urban Environment

The organisms inhabiting the urban areas influence the environment in many ways. Most of these impacts are positive and even necessary for the proper functioning of the urban ecosystem. These positive impacts are same to the role of the biota in natural ecosystems. The plants produce the organic matter and play an environment-forming role for many animal species. The animals feed on plants and transfer the organic matter. Soil dwelling animals form the normal soil structure and fertility of soil, fungi and bacteria decompose the litter and drive the main nutrient cycles. However, the anthropogenic activity in the urban environment can lead to significant changes in these natural processes. Many exotic plant species are intentionally introduced (Sjöman et al. 2016), but not all of them are suitable for the existing ecosystem. In example, the introduction of spruce (*Pīcea pūngens*) in the cities of steppe area in Russia leads to the degradation of grasses under the canopy, not only because of shading, but mainly because of acidification of soil. Landfills, then migrate to urban forests and recreational areas for an overnight stay and the soil surface at these areas becomes heavily polluted with bird droppings, including undigested pieces of plastic. The low pH also alters the solubility of heavy metals that become mobile and can enter the groundwater and the food chains. The animal species in urban areas can become a significant source of pollution. Large colonies of rooks (*Corvus frugilegus*) tend to feed on. Both examples represent a normal functioning of natural biota in unusual conditions. There is also another thing to be noted: the people, inhabiting the city and their domestic animals are also an important source of biological impact on the urban environment. The fecal pollution of urban water sources and in some cases of urban soils (due to drainage from the sewage system or due to the free-range domestic animals) is not uncommon in urban areas, especially in developing countries. An important issue is the epidemiological safety of the urban environment for the city inhabitants. Urban areas are the most densely populated and the polluted urban environment can be considered as a bioreactor with intensive horizontal gene transfer (Rizzo et al. 2013; Riber et al. 2014). Most plasmids, bearing the genes of antibiotic resistance have emerged in urban areas. Given the growing population density and the speed of spreading of the multiresistant strains of bacteria, it could lead to devastating outbreaks of diseases. Therefore, the biological impacts, while being largely ignored in urban studies, should be considered as a major threat to the sustainability of the urban ecosystem.

5 Monitoring and Bioindication as the Key Point for Developing the Sustainable Land Use Strategy

The assessment of the environmental quality in urban ecosystem is the key step to find the ways to overcome the existing problems and develop a sustainable land use strategy. The major hindrance is the complexity of the urban environment and a set of simultaneously acting negative factors that may affect the inhabitants of the city. Some of these factors are obvious, and can be measured directly. The others can remain unknown to the researcher. Furthermore, the interaction of different factors represents a new form of an impact, which often cannot be predicted.

The use of bioindication approach allows estimating the entire complex of negative influences, regardless of our knowledge about them. Bioindicators can also tell about the history of the problem, because the changes in communities can develop slowly, and reflect the time of the negative impact. The same is also true for long-living bioindicators, such as trees.

While the estimation of each factor via bioindication is usually inferior in accuracy to the direct measurement (i.e., chemical analysis), it may be much more informative for the final goal – the estimation of environmental quality in each particular case.

The principle of bioindication is the use of some biotic parameters to deliver the information on the changes in the environment. These parameters can scale from cellular level to the whole communities. On the cellular level, the changes of ultrastructure can be observed, with the genetic material being especially sensitive. In example, the anaphase analysis has been used to estimate the level of genotoxicity of urban soils (Gorbov et al. 2015). On the organism level, the changes can include the shifts in morphology (size of the organisms, teratogenicity), physiology and metabolism or behavior. The community-level bioindicators include the general biodiversity, species composition, the presence or absence of sensitive or tolerant species, which can help to indicate the nature of the negative impact.

Bioindication-based approaches can be divided into passive and active methods. Passive methods include the observation of existing communities and sampling the organisms from the environment to study them in the laboratory. The microbial and microinvertebrate bioindication is usually based on passive methods (Amossé et al. 2016; Pedrini-Martha et al. 2012). Active methods are based on placing the test organisms into the studied environment and detecting their response to the conditions. Active bioindication approach is usually based on plants that are planted in particular areas of the city.

Both approaches have strong and weak sides. The active bioindication is usually more accurate, because an appropriate control is always available. The bioindicators used share the same origin, and in case of plants, the same genotypes. However, it can only be used to estimate the "acute" environmental toxicity, present at the moment of introduction of the bioindicators. Nothing can be known about the time of the negative impact present. Passive bioindication can reflect the long-term effects of the negative factors; it is also more useful for the estimation of the local ecosystem stability, because the autochthonous organisms are studied.

The main problem emerging while using this approach is the proper control selection. It is particularly difficult for the urban studies due to high level of heterogeneity that is inherent to urban environment. For the urban soil studies it is even more difficult, because the areas in the city and outside its borders have different soil types. For the comprehensive estimation of the urban environment, we recommend to use both approaches to overcome the weaknesses of each method.

6 The Use of Plants as Bioindicators

Urban plants live in a highly unnatural environment. One of the most important environmental threats to the urban ecosystem is the atmospheric pollution created by anthropogenic activity. It is necessary to understand the ecosystem responses to the influence of urbanization in order to ensure that urban areas are well managed.

Plants used for monitoring environmental conditions are called bioindicators or biomonitors. Bioindicators can demonstrate the presence of air and soil pollutants and facilitate the estimate of the frequency of the occurrence of damaging levels. A bioindicator is any biological species (an "indicator species") or group of species whose function and population can reveal the qualitative status of the environmental conditions. Good biomonitor will indicate the presence of the pollutants and also attempt to provide additional information about the amount and intensity of the pollutants exposure.

A bioindicator can be defined as a vascular or nonvascular organism (in this case, plants) exhibiting a typical and verifiable response when exposed to a specific stressor, such as excessive pollution. These sensitive plants can be used to detect the presence of pollution at a specific location or region, which provides unique information regarding changes in air quality.

6.1 Characteristics of Bioindicators

Criteria for a suitable bioindicator are as follows:

- Plant should be easily found across wide geographic range
- It can grow in diverse habitats
- Easily recognized and has smaller-sized plants within its population
- Should have specific proven symptoms appear when exposed with pollutant
- It should display a consistent, increasing response

Several species have been extensively researched and are now available for planting as a bioindicator. Many highly pollutant-sensitive plant species have been evaluated for their potential use as a bioindicator species capable of detecting the presence of ozone air pollution through the development of very specific and distinctive foliar symptoms. Among the various categories, air pollution by automobiles is the most insidious one, which exerts highly detrimental effects on living organisms, vehicles releasing large quantities of pollutants such as oxides of nitrogen, sulfur, carbon, heavy metals, dust, and particulate matter.

Hijano et al. (2005) research shows that the coniferous species such as *Pinus pinea*, were more sensitive to SO_2 atmospheric concentration than leafy species as *Quercus ilex* subspecies *ballota* and, in the same way, bush species, such as *Pyracantha coccinea* and *Nerium oleander*, and were more sensitive than woody species, such as *Cedrus deodara* and *Pinus pinea* respectively.

6.2 Potential Bioindicator Plants

Forbs: *Centaurea nigra* and *Impatiens parviflora*.

Shrubs and herbaceous plants: *Alnus incana*, *Corylus avellana*, and *Sambucus racemosa*, *Rubus* sp., *Apocynum cannabinum, Aster macrophyllus*, *Apocynum cannabinum*, *Rumex patientia* L., *Viburnum tinus,* and *Sambucus nigra.*

Trees: *Prunus serotina*, *Liriodendron tulipifera*, *Fraxinus americana*, *Sassafras albidum, Tilia americana*, *Platanus occidentalis*, *Salix herbacea*, *Fagus* sp., *Prunus pensylvanica.*

Roadside plants: *Bougainvillea spectabilis*, *Ageratum conyzoides, Ficus religiosa*, *Cynodon dactylon*, *Peltophorum pterocarpum*, *Portulaca oleracea*, *Ricinus communis*, *Bambusa bambos*, and *Terminalia catappa*.

Roadside plants in urban areas demonstrate wide responses when exposed to atmospheric pollutants in the form of respiration, photosynthesis, enzymatic reactions, stomatal behavior, membrane disruption, senescence, and ultimately death (Rai and Panda 2015). Hydrangeas are a good example for bioindicator plant, however Al is necessary for blue flowers, in alkaline soils, there may be Al deficiency, which result in pink *Hydrangea* flowers. Algae can also be a good indicator of water quality because they react rapidly to changes in levels of N and P. Lichens live on surfaces such as trees or rocks or soil and are very sensitive to toxins in the air. Black poplar (*Populus nigra* L.) is a good indicator for ground level ozone and high sulfur dioxide in air damages coniferous trees.

7 Microbial Bioindication in Urban Areas

The sustainability of the entire urban ecosystem functioning is based on the properties of the soils, which are influenced by the resident microbial communities. These communities can be used to extend the number of bioindicators applicable for the monitoring of urban environment.

The assessment of microbial communities has a number of potential advantages. First, the microbial populations can react rapidly on the changes in the environment. Secondly, the microbial communities are quite sensitive and even small doses of contaminants can lead to sharp population decrease or even eradication of some species. The reaction of microbial communities to the environmental stress can include shifts in numbers of not only particular species, but even whole functional groups within the population, the overall loss of biodiversity or the changes in the biochemical activity of microbial community.

The main challenges associated with microbial indicators root from the same properties of bacteria that lead to the advantages. The ability of fast reproduction that is inherent to the microbes can lead to high temporal and spatial variability of microbial populations, which is particularly notable in heterogeneous urban environments. The accurate identification of the microbes can also be quite difficult and time-consuming. The possible ways to overcome these problems include the search for stable microbial indicators and reference microorganisms that are identified easily and react to specific types of impact. Another possible solution is the determination of microbial communities' characteristics for different land-use types within the city. Therefore, the development of new approaches to microbial bioindication in urban environment is a challenge that needs joint efforts from microbiologists and urban ecologists. The next subsections are aimed to summarize the experience that exists to date in that field together with presenting some observations made by the authors.

7.1 Microbial Biomass

Microbial biomass and biomass-related indicators are widely used in the assessment of the community reaction to the stress. Microbial biomass reflects the total amount of the microbial community, and is usually reduced if the soil is contaminated with hydrocarbons (Lorenz and Kandeler 2005; Megharaj et al. 2000). Nevertheless, the microbial biomass should be used with caution, since conflicting data exist on its dynamics at pollution. In particular, the microbial biomass is not a reliable indicator of soil pollution with heavy metals, as different studies have demonstrated an increase in microbial biomass upon the addition of a mixture of metals to the soil and decrease upon the use of silt contaminated with heavy metals as a fertilizer (Gil-Sotres et al. 2005). Some indicators related to the microbial biomass have proven to be reliable in urban studies. It has been shown recently that microbial biomass nitrogen can serve as important soil health indicator to predict soil quality and productivity in highly disturbed soils in urban areas (Knight et al. 2013). The existing controversy in the direction of microbial biomass changes under stressful conditions may be due to the lack of unified method of its determination. The methods include various modifications of chloroform fumigation-extraction method (Brookes et al. 1985), microwave-irradiation (Islam and Weil 1998), indirect estimation based on basal respiration measurements (Anderson and Domsch 1978), and direct estimations based on cell counts and recalculation. Given that the soil is an extremely complex environment, even slight modifications of each of these methods can lead to achievement of results that are hardly comparable to those obtained by other researchers. The results obtained by the methods that rely on different principles are even more difficult to compare. The actual microbial community includes active, potentially active, dormant and dead cells (Blagodatskaya and Kuzyakov 2013). The degree of dormancy can also vary greatly. Some cells are still living, but metabolically inactive—such microorganisms would contribute to biomass, determined via microscopic analysis, but would not to respiration-based methods. The physiological state of the microorganisms is also very important: the starving cells can have lower protein content, thus affecting the microbial biomass C and N parameters. To conclude, for accurate estimation of microbial biomass in soil, at least two different methods should be used simultaneously.

7.2 Changes in Soil Respiration, Carbon Cycling, and CO_2 Emission

The most important function of both natural and urban soil is the biogeochemical cycling of elements. The global carbon cycle includes the processes of carbon release into the atmosphere via natural biochemical processes (fermentation, respiration) and anthropogenic activities (fuel combustion). The opposite part of the cycle is CO_2 fixation by autotrophic organisms (photosynthesis and chemosynthesis) and subsequent carbon deposition in the form of organic matter or sedimentation as calcareous rocks. Soil organic matter is a major pool of biogenic carbon, which is estimated to be over 1500 billion tons (Lal 2004). The soil organic matter plays an essential role in soil structure formation, its functioning and productivity. The soil organic carbon pool exists as an equilibrium between gains and losses and the increased rates of decomposition of soil organic matter can lead both to soil degradation and extensive emission of CO_2. Urban areas have been characterized both by higher SOC densities in comparison to rural areas (Pouyat et al. 2002) and by lower (Jo 2002) depending on the climate in the studied cities. However, the difference is rather not quantitative, but qualitative. Urban soils are subjected to specific pedogenic processes with a leading anthropogenic factor, such as mixing, burying, sealing, etc. Such processes modify the quantity, quality, and depth distribution of substrates for decomposition; the microbial communities of decomposers are also altered, as well as the diffusive transport through soil profile (Lorenz and Lal 2009). It is clear that a better understanding of the urban soil properties is urgently needed in the context of their role in the global carbon cycle as well as their local scale ecosystem-maintaining role.

The studies of soil respiration are most frequently used to assess the soil quality and the state of the microbial community dwelling in it. The basal respiration has been shown to be sensitive to heavy metal pollution (Gülser and Erdoğan 2008), decreasing with the growing level of pollution. However, the metabolic quotient, qCO_2 is generally considered a more reliable indicator. This quotient serves as an estimation of metabolically active portion of microbial community and is calculated as basal respiration to microbial biomass. It has been shown that higher qCO_2 values are associated with younger urban landscapes gradually decreasing to the older areas of the city. This suggests that the gradual decrease of the metabolic quotient value reflects the succession in urban landscape with older areas of the city closer to steady-state conditions (Scharenbroch et al. 2005). The highest qCO_2 values have been found in urban lawns, which are considered to be caused by land management and disturbing impacts on soil microbial community (Vasenev et al. 2015). Recent studies have shown that in a complex urban environment edge effect should be taken into account when considering soil respiration and carbon cycling. The respiration rates have been shown to be highest at the edge of sealed areas, gradually decreasing in the direction of the green area interior (Wu et al. 2016).

Another valuable tool in respiration-based bioindication studies is the method of substrate-induced respiration (SIR). The method allows to estimate both general microbial activity in soil, by using glucose as a substrate, and to study more specific changes in microbial community response to environmental stress by using a set of different substrates and obtaining catabolic profiles. It has been shown that the tolerance of SIR to Pb contamination was concomitant with deep changes in catabolic profiles in these soils (Bérard et al. 2016). It may be possible to establish a set of indicative patterns of catabolic profile changes for different soil and land-use types which makes SIR a valuable instrument for microbial bioindication in urban environment.

7.3 Changes in Nitrogen Cycling and N_2O Emission

The global nitrogen cycle is the second largest biochemical cycle after the carbon cycle. The natural nitrogen cycling processes are greatly affected by anthropogenic activities. These impacts include inputs of reactive nitrogen species derived from fossil fuel combustion, application of the chemical fertilizers and transformation of the environment (Pierre et al. 2016). The nitrogen cycling includes the process of nitrogen fixation, leading to incorporation of chemically inert dinitrogen gas into bioavailable compounds entering the food chains. The opposite process is nitrogen mineralization, where organic N is converted to ammonium (NH_4^+) and then to nitrate (NO_3^-). The latter is reduced to N_2O via microbial denitrification process, and finally to N_2. Nitrous oxide is an important greenhouse gas, and the urban land-use has been shown to alter N cycling rates and N_2O fluxes to the atmosphere.

Nitrogen cycling has been poorly characterized in urban areas (Zhu et al. 2005). Still, there is some evidence that nitrogen-cycling parameters are affected by urban impacts and therefore can be used in bioindication. Various activities have been studied in relation to nitrogen cycling in soils. The proteolytic activity has been shown to have no coherence with numbers of proteolytic bacteria of site-specific properties (Bach and Munch 2000). This is possibly due to the fact that proteolytic activity changes are caused by differences in expression rates and not by the changes in microbial population. Moreover, the exoenzymes exist in soil independently from the bacteria that have produced those (Schloter et al. 2003). This leads to a slower response of this parameter to changing conditions. However, a deep decrease of proteolytic activity may indicate a long-lasting negative impact on the microbial community.

One of the most sensitive stages of the nitrogen cycle is nitrification. It has been shown that nitrification is altered by several soil properties, such as pH values, soil organic matter, and heavy metal content. The sensitivity of nitrification to a set of environmental properties makes it quite difficult to interpret the results of testing. Thus this process should not be considered as a straightforward bioindicator (Sauvé et al. 1999).

Nitrogen fixation has been shown to be also very sensitive to anthropogenic impacts (Filip 2002). In urban soils, the total nitrogen income is higher than in natural environments due to abundance of artificial sources of reactive nitrogen species. This can possibly play a role in repression of biological nitrogen fixation, as the presence of nitrate is well known to inhibit the nitrogenase activity (Cejudo and Paneque 1986). Therefore, the presence of reactive nitrogen species in soil samples should be taken into account when using direct measuring of nitrogen-fixing activity (i.e., via acetylene reduction technique) as a bioindicator.

The process of denitrification has been widely discussed in literature due to the role of this process in the emission of N_2O, an important greenhouse gas (Raciti et al. 2011). However, little attention has been paid so far to the use of this process for bioindication purposes, especially in urban environment. The quantity of denitrifying bacteria has been placed among most sensitive parameters (Filip 2002) indicating the anthropogenic impact. Our findings (Gorovtsov et al. 2013) have shown that the numbers of denitrifying bacteria, determined by MPN method varied greatly between the sampling sites from as high as 10^7 cells/g to as low as 10^2 cells/g of soil. The highest numbers were found in the sampling site close to the market entrance, where the soil was amended with food waste, and the lowest at the street side with degraded grass cover (bare soil), indicating the importance of available carbon sources for denitrification.

The nitrogen cycling features of urban soil remain to be poorly studied from the point of bioindication, and further research is needed to establish reliable parameters for practical use.

7.4 Changes in Microbial Abundance and Community Structure

The microbial communities react to the anthropogenic impacts by changes in their quantitative and qualitative characteristics. The decline in numbers of microorganisms, caused by the initial impact can be restored after some time by the adaptation of the community and growth of tolerant strains of bacteria which occupy the vacant ecological niches. Thus, the estimation of the numbers of bacteria in soils is insufficient to estimate the level of the environmental stress.

The changes in species composition and proportions of different groups of bacteria can be obvious even at the initial stages of the study. In our studies, the simple plating of soil dilutions on the nutrient agar revealed the deep differences in soil microbial communities in the city of Rostov-on-Don (Fig. 9.2).

The soils of the urban areas in Rostov-on-Don were rich in culturable bacteria forming pigmented colonies of yellow, orange, pink, and red color, which were absent in relatively undisturbed soil of the old fallow field adjacent to the city. The pigmented microorganisms were later identified with MALDI-TOF Biotyper, Bruker Daltonics, as members of *Microbacterium*, *Arthrobacter,* and *Rhodococcus* genera. The strains belonging to *Arthrobacter* sp. were dominant in the soils of the central parts of the city.

The proportion of *Bacillus* sp. gradually decreased in the urban–rural gradient toward the city center. The proportion of *Bacillus* spores in the culturable part of the

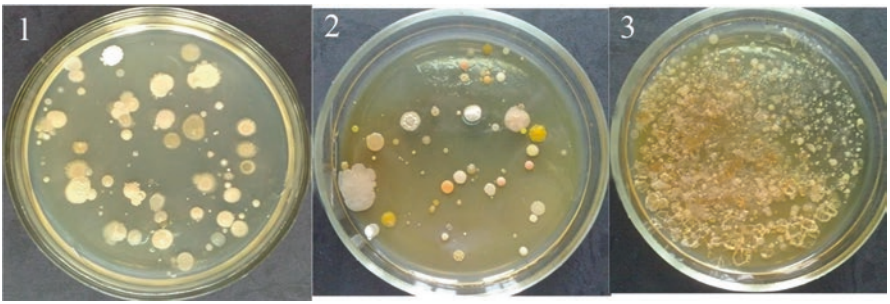


Fig. 9.2 The colonies of culturable bacteria on nutrient agar obtained from undisturbed soil (*1*), soil from city outskirts (*2*), soil from city center (*3*)

microbial community was determined by plating the same soil suspension before and after pasteurization. In undisturbed soils, the *Bacillus* spores comprised at average 4–5% of the bacteria, growing on nutrient agar, reaching a maximum of 25% at the plot of virgin steppe outside the city. In soils of city outskirts the share never exceeded 3–4%. In the soils of the city center, a sharp and statistically significant decline was observed with the proportion of *Bacillus* sp. spores as low as 0.1–0.01%. The proportion was greatly dependent from the state of grass cover, being the lowest at the plots with bare soil. It was also higher in autumn than in summer and spring. These findings indicate that the leading factor for the decrease in proportion of *Bacillus* sp. in urban soils is the availability of fresh organic matter from plant residues. The prevalence of *Arthrobacter* and *Rhodococcus* species in the soils of city center may be indicative of soil pollution with organic compounds, such as PAHs and petroleum hydrocarbons. Based on our observations, we therefore recommend to use the proportion of *Bacillus* sp. in the microbial community as a bioindicator of urban soil quality.

The reaction of various groups in the microbial community to specific pollutants has been reported in literature. It has been shown that cultivable *Pseudomonas* spp. increased its abundance upon the treatment of soil with PAHs (Niepceron et al. 2013).

There are also studies indicating that the contamination with heavy metals have various adverse effects on the microbial community (Lenart-Boroń et al. 2014). It has been shown that bacterial community was more sensitive to contamination with Zn and Cu than fungal community, which led to an increase in fungal/bacterial ratio with increasing level of contamination. However, the lower pH values enhanced the negative effect for bacteria, but not for fungi (Rajapaksha et al. 2004). Juwarkar et al. (2007) have shown the decrease in abundance of several groups within the microbial community for cadmium and lead contaminated soils. The decrease was most significant for actinomycetes and diazotrophic bacteria of the genus *Azotobacter*, indicating the possible use of these groups in bioindication of contaminated soils. Lenart and Wolny-Kaładka (2013) have shown that in the urban soils of Cracow with polymetallic contamination all groups of soil microorganisms reduced their numbers compared to uncontaminated soil.

Thus, the quantity of certain groups of bacteria and the taxonomic structure of the microbial community is a valuable tool in bioindication. The development of molecular methods and culture-independent approaches will contribute greatly to the search for bioindicator organisms by the inclusion of unculturable groups of bacteria that are abundant in soil environment.

7.5 Changes in Soil Enzymatic Activity

Though the quantity of particular groups of bacteria may be a useful parameter in soil studies, it is not always directly linked to the soil quality. This is due to the fact that the proper functioning of the soil is dependent on the biochemical processes that are mediated by different enzymes. The enzymatic activities in soil are highly dependent on the physiological state of the microorganisms that produce these enzymes, which can alter the levels of expression of these proteins. For instance, if the majority of the cells are in dormant state due to unfavorable conditions, the enzymatic activity can be very low, while the direct counts, plate counts or qPCR-based estimations can still show large numbers of bacteria present in the sample. Another important factor is the influence of soil conditions. The activity of certain soil enzymes, especially exoenzymes, can be altered by soil pH, the heavy metal ions disrupting the disulfide bonds, nonspecific influence of soil organic matter, etc. Thus, even the sufficient level of expression in soil bacteria does not guarantee the high level of soil performance in the ecosystem. Still, the study of the soil enzymatic activity is irreplaceable for understanding of the soil functioning and the mechanisms of negative anthropogenic impacts.

There is a plenty of enzymes that can be studied for their activity in soil and many of them have a potential to be used in bioindication. These enzymes can be grouped according to the biogeochemical cycles they are involved into.

The enzymes involved in the carbon cycle include mostly hydrolases that are used to decompose polysaccharides of the plant residues. This group include amylases, α-glucosidase, β-glucosidase, 1,4-β-cellobiosidase, β-xylosidase, polyphenol oxidase, and invertase.

Among these, β-glucosidase is used most widely, as it is considered to indicate the soil quality and is linked to the SOM quantity and quality (de Almeida et al. 2015).

The nitrogen cycle is represented by urease, N-acetyl-β-D-glucosaminidase, total protease activity or more specifically, arginine aminopeptidase and tyrosine aminopeptidase activities. These enzymes play a crucial role in nitrogen mineralization, decomposing proteins, chitin and urea. The most commonly studied enzyme in this group is urease, due to the simplicity of the determination and high sensitivity to various negative impacts. Li et al. (2015) observed a significant decrease in urease activity in roadside soils in Beijing.

Phosphorous cycling enzymes are represented mainly by phosphatases with different optimal pH levels. Most commonly, the activity of alkaline (Wang et al. 2007) and acid phosphatases (Wieczorek et al. 2014) are measured, but some studies also include the phosphatase activity at neutral pH values (Cui et al. 2013). Due to high sensitivity of these enzymes, phosphatase activity is one of the most popular parameters in soil ecotoxicological studies, second only to dehydrogenase activity.

Among the sulfur cycle enzymes, the arylsulfatase activity is studied most extensively. This enzyme has been shown to be particularly sensitive to zinc, copper, and nickel contamination (Wyszkowska et al. 2016).

There is also a group of enzymes that are not directly linked to any biogeochemical cycle but play a crucial role in microbial metabolism. This group of parameters includes, for instance, catalase and dehydrogenase activities. The latter is the most widely used parameter in soil enzymology. It should be stressed that the enzymes of this group can be used for the estimation of the general state of the soil microbial community, while the enzymes involved in biogeochemical cycles of elements can reveal its more specific traits.

It is also notable that the vast majority of the element-cycling related enzymes belong to the hydrolase class. They are considered mainly from the point of mineralization of the corresponding elements for plant nutrition. This approach roots from the early studies, the majority of which was dedicated to agricultural soils.

Far too less attention is paid to the synthesis of the soil organic matter, which is a critical step for sustaining the soil quality in human-affected environments. Thus, while the up mentioned enzymes have proven to be useful tools for bioindication, there is still an urgent need to develop additional parameters for soil quality assessment.

8 Conclusions

Bioindication is a valuable instrument for the environmental quality assessment. It allows estimating the entire complex of negative factors and predicting their impact on the living organisms. However, there is no method that can allow making an accurate estimation, taken separately. We recommend using both active and passive bioindication to benefit from the strong sides of each approach.

We also recommend choosing carefully among the great number of parameters employed previously in bioindication studies. In general, all the parameters can be divided into two groups: those that characterize the whole community (i.e., microbial biomass, soil respiration, total dehydrogenase activity), and those that measure some specific features or processes (numbers of particular groups of bacteria, activity of enzymes with high substrate specificity, catabolic patterns). For the impacts that apparently influence a wide range of organisms, the choice of the parameters from the first group is more preferable. The parameters from the second group may be very sensitive, but they characterize only some part of the microbial community and should be used in the studies of particular soil functions or properties. In other words, a set of the parameters for bioindication should be consistent with the aim and scope of each study—there are no universal solutions.

The methods of bioindication of the environmental quality are developing rapidly, but there is still much to do in this field. Some parameters (like microbial biomass) remain controversial. Further research is needed to establish reliable parameters to study the element cycling, especially in urban environment. There is also a need for generally accepted list of microbial and plant bioindicators, which would contribute to our knowledge by comparative studies in different conditions.

In our view, the main future prospective in the field of bioindication is the transition from random selection to a common system of indicators, with the use of standard methods, based on a comprehensive approach to the studies of environmental quality.

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