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32 *Nonlinearity of an Ecosystem Response to Toxic Load: a Fundamental for Environmental Quality Estimation*

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ABSTRACT The biota response to a toxic load is significantly nonlinear (i.e. an ecosystem responds as a trigger, according to the law "everything or nothing"): an ecosystem is in a normal state until the threshold of the load is exceeded; when it is exceeded a small increase of the load causes a disproportionally strong response of the ecosystem which very rapidly jumps into a maximally disturbed state. This nonlinearity has several important consequences for environmental quality evaluation. It is possible to categorize a response of certain ecosystem component into only three gradations (normal, buffer, and impact). This strongly restricts the resolving power of the quality estimation (only rough estimation is possible). The biota parameters give quite different information compared to the pollution data (if the dose-response relationships are linear, the biota indicators would give the same information but in other units). The stage of ecosystem transformation can be used as a highly aggregated index of environmental quality.

KEYWORDS Environmental quality, dose-response relationship, nonlinearity, air pollution, natural ecosystem, biological indicator

1. INTRODUCTION

Usually the most simple questions are the most difficult ones to answer. Among such difficult simple questions are "what is the environmental quality?" and "how can we measure it?". A great number of articles on environmental assessment are basically attempts to answer these questions from various points of view. In this article one of the least developed problems in environmental quality is addressed: the quality of natural ecosystems (i.e. any ecosystem except artificial ones such as arable, urban, industrial, etc.).

As a working definition consider *natural ecosystems quality* (NEQ) as the degree of fulfilment of humanly important functions by an ecosystem. These

functions are: (1) fulfilment of people's requirements in utilization of biological resources; (2) satisfaction of aesthetic and recreational needs, and the needs for a healthy environment (where people may stay without risking their health); (3) support of landscape stability; and (4) contributing to the functioning of ecosystems of higher rank, up to the biosphere. With such understanding the NEQ is primarily determined by the biota (by its structure and functioning intensity).

This article will not consider the possible difference in quality of various ecosystem types (e.g. a mature stand and a saline land), but will focus on the way the NEQ of certain ecosystem types alters under anthropic pressure. The empirical data considered concern airborne pollution of terrestrial ecosystems on a local scale (the minimal space unit is a catch land, the total area of about 1–3 thousand km^2). However, the conclusions will be general.

2. FORMULATION OF THE PROBLEM

Most methods of environmental quality assessment using biological indicators (such as the De Sloover-LeBlanc index of atmosphere purity, the Pantle-Buck saprobic index, etc.) are based upon the assumption that increase of the environment pollution (even if not very strong) causes more or less proportional changes in the condition of indicators and, correspondingly, environmental indices. The relationship "*indicator (index) vs. pollution*" is thus linear. The main issue which will be discussed here is whether this assumption is correct and, if not, how this affects environmental quality assessment.

A good model object for checking the validity of various NEQ evaluation methods (including environmental indices) is a territory surrounding a powerful long working point source of pollutant emission. Approaching the source of pollutant emission gradually one may see by the naked eye how the NEQ alters (usually it aggravates). Correspondingly, it is easy to evaluate the results of different evaluation methods.

There are many investigations of an ecosystem response to pollution from a point source of emission. But most of these have the same disadvantage, in that they are based on pair comparisons where ecosystem parameters in the most polluted place (the "test") are compared with unpolluted ecosystem parameters (the "control"). Such an approach allows us to analyze only a general trend of changes and its maximal amplitude, and does not allow us to analyze the form of trajectory of an ecosystem response to the toxic load. This aspect is very important for NEQ evaluation.

In classical toxicology the action of poisons on an organism is analyzed on the basis of the *dose-response* relationship (or *concentration-active curve*). The aim of the analysis of natural ecosystems response trajectories is actually to construct similar relationships at the level of a whole ecosystem, rather than a separate organism. In this case a dose is a toxic load value which may be estimated from the deposition or accumulation of pollutants in the ecosystem, and an effect is a set of parameters, completely describing the condition of the ecosystem.

This problem was formulated in the mid-1970s (see, for example, Fedorov 1976) in connection with the development of ecotoxicology. However, solving it in

practice is difficult because it requires sintopic registration of pollution levels and a large number of biota parameters of genetically uniform ecosystems on many sample sites uniformly located along the pollution's gradient. Consequently there have been relatively few attempts to build dose-response relationships at an ecosystem level (Freedman and Hutchinson 1980; Nordgren, Baath, and Söderström 1985; Stepanov 1995; 1992; Trubina and Mahnev 1997; Armand et al. 1991; Tsvetkov 1993; Alexeyev and Tarasov 1990; Saliev 1988).

3. THE METHODICAL APPROACH

For the analysis of *dose-response* relationships at ecosystem level results on responses of a southern taiga forest ecosystem on airborne pollution from a copper smelter in the Middle Ural (near Revda, 50 km west of Ekaterinburg) will be used. This smelter has operated since 1940 and emits SO_2 (130000 ton/year), Cu (2600 ton/year), Pb (560 ton/year), As (640 ton/year), Cd, Zn, and other elements. Pollution with polymetallic dust combined with additional acidification of soils which are naturally acidic leads to dramatic degradation of forest ecosystems. The territory pollution and ecosystem transformation have been described previously in detail (Vorobeichik, Sadykov, and Farafontov 1994; Kaigorodova and Vorobeichik 1996).

The general methodical approach to construct dose-response relationships at ecosystem level is the following. Approximately 30–80 sample sites are situated at various distances from the source of emission (usually such a quantity is enough to reveal the pollution gradient and to correctly approximate the dose-response relationship by nonlinear regression). The toxic load and biota parameters are measured at each sample site.

The toxic load is evaluated from the levels of predominating pollutants in natural absorbic media (snow, upper soil layer, etc.). In the case under discussion the main pollutants from the copper smelter – Cu, Pb, and Cd, 5% HNO_3 – were selected, and extractable forms were measured in the upper 0–5 cm soil layer. To reduce the pollution data the following toxic load index was used:

$$D_i = \frac{d_i}{\min\limits_i (d_i)}, \; d_i = \sum_{j=1}^{k} \frac{X_{ij}}{\min\limits_i (X_{ij})},$$

where X_{ij} is the concentration of metal j ($j=1,...,k$) at sample site i ($i=1,...,n$). The index is measured in arbitrary units and shows how many times the background level at a particular site is on average exceeded by all pollutants. Strictly speaking, the index has no toxicological sense and serves only as a marker of the whole pollution complex (including even those pollutants which were not measured).

A biota is characterized by about 120 parameters embracing a timber stand, a herb layer, forest litter, soil macrofauna, soil enzymes, and epiphytic lichenosynusia. The list of the main biota parameters measured is given in Table 32.1. The measurement methods are described in detail in the original articles (Vorobeichik, Sadykov,

TABLE 32.1 *Investigated biota parameters.*

Component	N_1	N_2	List of the main parameters
Timber stand	28	40	Standing crop; density of stand; fullness of site; tree height; diameter of trunk; degree of sheltering; portion of dead tree; density of regeneration.
Herb layer	28	25	Number of species: grasses, short herbs, tall herbs, etc.; portion of biomass: grasses, short herbs, tall herbs, explerents, forest species, etc.
Forest litter	19–35	12	Average thickness; range of thickness.
Soil macrofauna	26	18	Total abundance; abundance of Lumbricidae, Enchytraidae, Carabidae, Lithobiidae, Elateridae, etc.; portion of saprophages, herbivores, carnivores.
Soil enzymes	18–88	8	Activity of cellulase, phosphatase, etc.; rate of cellulose decomposition; rate of nitrogen fixation.
Epiphytic lichenosynusia	13–29	15	Number of species; cover on trunk base; cover at a height of 1.3 m; the height of climbing.

N_1 = amount of sample sites; N_2 = total number of parameters.

and Farafontov 1994; Vorobeichik 1995; 1998; Vorobeichik and Khantemirova 1994; Mikhailova and Vorobeichik 1995).

The relationships *biota parameter vs. toxic load index* was approximated by the logistic equation:

$$y = \frac{A - a_0}{1 = e^{\alpha + \beta X}} + a_0,$$

where y is a biota parameter, x is the index of the toxic load, and α, β, a_0, and A are coefficients. The coefficients of the equation were calculated from Marquard numerical assessment.

One benefit of a logistic equation is that it is analytically easy to obtain the coordinates of critical points (points of inflection) by analysis of derived functions. Important information is obtained from three critical points – upper, middle, and lower, corresponding to the beginning, the middle, and the end of the "rapid" parameter alterations. The abscissas of these points are equal:

$$X_U = \frac{-\alpha + \ln(2 - \sqrt{3})}{\beta}, X_M = -\frac{\alpha}{\beta}, X_D = \frac{-\alpha + \ln(2 + \sqrt{3})}{\beta}.$$

The difference between the abscissas of the upper and the lower critical points is a part of the pollution gradient where "rapid" parameter alterations occur. If such

a part is small in comparison with the whole gradient length, the nonlinearity of the response is very significant.

4. NONLINEARITY OF ECOSYSTEM RESPONSE TO TOXIC LOAD

Examples of dose-response relationships are shown in Figure 32.1. Firstly it may be noted that the curves have a well expressed S shape. This indicates significant nonlinearity of the biota response on a toxic load: an ecosystem is in a normal (nondisturbed) state until the threshold of the load is exceeded, after which point a small increase in the load causes an unproportionally strong response of the ecosystem which very rapidly jumps into a maximally disturbed state. "Rapid" changes being over, any further increase of the toxic load does not cause significant change. In other words, an ecosystem responds to toxic load as a trigger, according to the law "everything or nothing".

This activity was observed in most analyzed biota parameters: over 80% of them are approximated by logistic equations satisfactorily (determination coefficients are higher than 30%) (Figure 32.2(a)). In almost half of the biota parameters the nonlinearity is very strong: "rapid" changes occur in a 5% portion

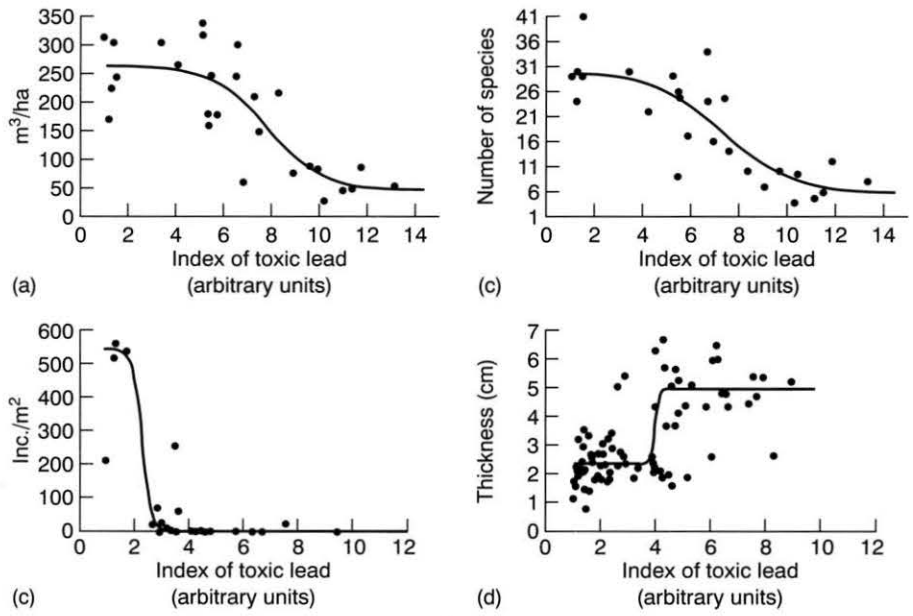

FIGURE 32.1 *Relationship biota parameter vs. index of toxic load: (a) standing crop* (R^2 = *0.681); (b) diversity of herb layer* (R^2 = *0.698); (c) Earthworm abundance* (R^2 = 0.923); (d) forest litter thickness (R^2 = *0.648).*

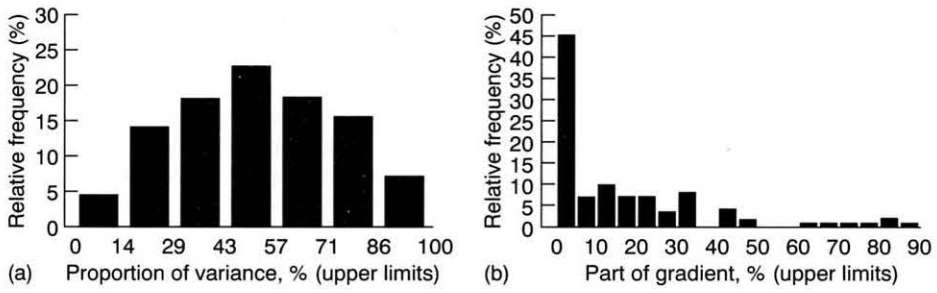

FIGURE 32.2 *Frequency distribution of characteristics of dose-response relationships for a set of biota parameters: (a) the value of variance explained by logistic equations; (b) the portion of the pollution gradient with "rapid" change.*

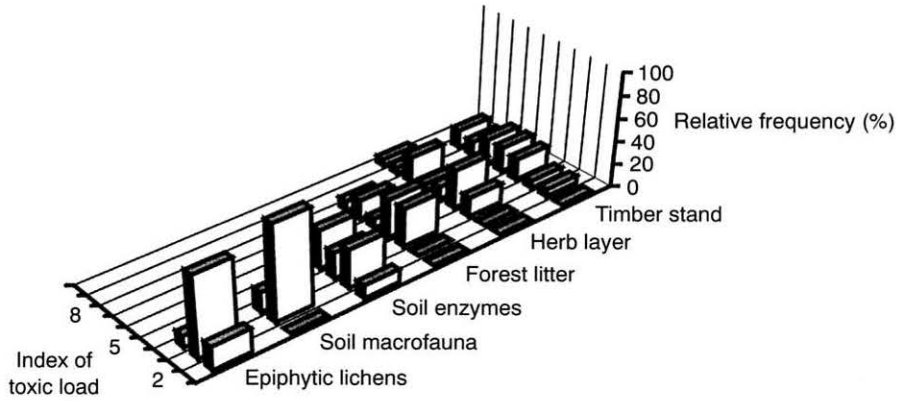

FIGURE 32.3 *Frequency distribution of abscissas of the upper critical points for various ecosystem components.*

of the gradient (Figure 32.2(b)). It is important that response thresholds of different biota parameters (abscissas of the upper critical points) are distributed along pollution gradient not uniformly but as clusters (Figure 32.3). There are groups of parameters with earlier (epiphytic lichenosynusia soil macrofauna, soil enzymes) and later (timber stand, herb layer) responses. In other words, the response trajectories of the biota parameters are "tied" in reasonable wide bundles (a group of parameters responding as one parameter).

So the response to the earlier question "is the response of bioindicators to pollution linear?" is definitely "No". Similar results were obtained by other investigators in other natural conditions (Nordgren, Baath, and Söderström 1985; Stepanov 1995; 1992; Trubina and Mahnev 1997; Armand et al. 1991). This allows us to say that nonlinear ecosystem responses are general and predictive.

5. LINEAR MENTALITY AGAINST NONLINEAR NATURE

An S-curve form of the dose-response relationships is an axiom in classic toxicology. It is also a graphic form of expressing the Shelford tolerance law, one of the principles in autecology. However, when analyzing data on natural ecosystems the biota parameter vs. pollution relationship may often be approximated as a straight line. This is primarily connected with the above mentioned peculiarity of investigations on polluted territories, as when there are only two or three sample sites (test and control) it is difficult to obtain anything other than a straight line. But even if there are many sample sites, investigators often approximate the dose-response relationship as linear. Figure 32.4 illustrates such situations when linear mentality makes nonlinear dose-response relationships linear.

The first example is an early analysis by Tyler (1974) of the microbe community in the vicinity of a copper smelter in Sweden (nonlinearity is excluded by logarithmic transformation). The second example is the recent investigation of Australian authors (Yeates, Orchard, and Speir 1995) on the effect of heavy metals on earthworms. The third example is the recent work of Russian explorers (Moiseenko and Kudrjavceva 1995) aimed to study the action of Ni pollution on fish. In all cases the nonlinear approximation by a logistic equation which was made based on the author's data described the relationship with more accuracy than the linear approximation made by the authors. In the last two examples the benefit of accuracy is very significant.

There are many other such examples, but these are enough to conclude that linear mentality is international and resistant in time. The main reason for the viability of the linear mentality method is that it is much easier to describe responses in this way (all handbooks on statistics advise eliminating nonlinearity using various methods of linearization). However, this simplification may be accepted only as a first approximation too rough for the modern level of knowledge on the regularities of ecosystem response.

6. LINEAR AND NONLINEAR MODELS: DIFFERENCES IN APPROACHES OF ENVIRONMENTAL QUALITY EVALUATION

Another natural question at this point is: "Is it important how we approximate the dose-response relationship (by regression)? Is it not a mere manifestation of mathematical rigorism?" The approaches of NEQ evaluation based on the linear or nonlinear model are cardinally different. Table 32.2 shows these differences.

The following comments are necessary. If we admit that the relationship *biota parameter vs. toxic load* is linear, then measuring a certain biota parameter we can uniquely (within the measurement error) predict the pollution level and the value of a certain biota parameter. Conversely, if we know the pollution level we can uniquely predict the value of any biota parameter. In this case the biota parameters give quite different information on the NEQ from that given by the pollution data. They only express this information in other units of measurement, the transition

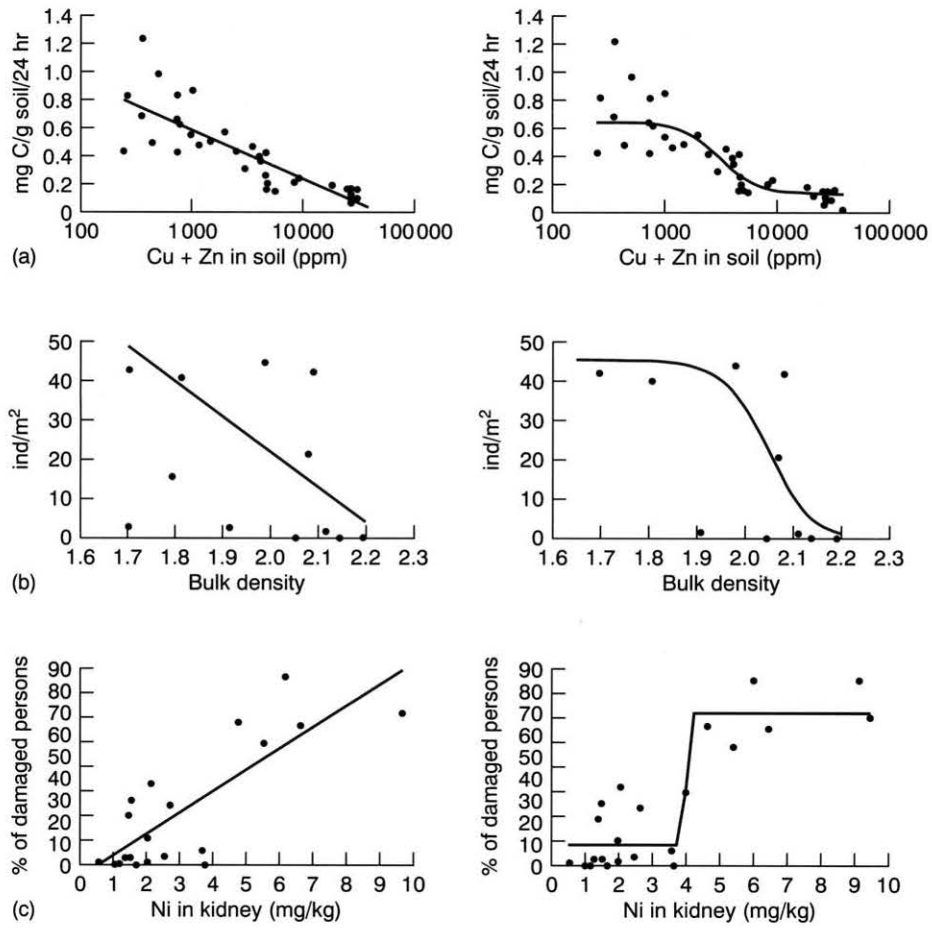

FIGURE 32.4 *Examples of "linear mentality" (left – author's approximation by a straight line, right – approximation by a logistic equation based on the author's data): (a) soil respiration vs. soil pollution (Tyler 1974) (left – R^2 = 0.7144, right – R^2 = 0.7183); (b) earthworm abundance vs. soil bulk density (Yeates, Orchard and Speir 1995) (left – R^2 = 0.4108, right – R^2 = 0.6406); (c) nephrocalcitosis cases in fishes vs. Ni levels in kidneys (Moiseenko and Kudrjavceva 1995) (left – R^2 = 0.7278, right – R^2 = 0.8426).*

between which is unambiguous (e.g., as temperature measurement in Celsius or Fahrenheit scales). In other words, if the linear model is correct, biota parameters are a convenient supplementary instrument to measure the environmental quality (e.g., easy or low-cost measurements), but an unnecessary one. The most important point is that if we accept the linear model then we equalize the NEQ and the pollution level.

TABLE 32.2 *Differences between linear and nonlinear models for the environmental quality evaluation.*

Comparison criteria	Linear model	Nonlinear model
1. Are biota parameter data and pollution data the same data?	Yes (but units are different)	No (they give quite different information because threshold of indicators are unknown a priori)
2. Are there any differences among certain biota parameters data?	No	Yes (differences are great because threshold values of individual indicators are different a priori)
3. Predictability of the environment quality using pollution data	Good	Poor
4. Scale for environment quality measurement	Ratio scale	Ordinal scale
5. Resolving the power of environment quality evaluation	High (10^1–10^2 gradations)	Low (10^0–10^1 gradations)
6. Benefits of the use of biota parameters compared to pollution data	Easier, lower-costs and higher measurement accuracy	Biota parameters are the environment quality evaluation as such, pollution data are quality predictors

The situation is different if we admit that the relationship is nonlinear. Firstly, it is not known a priori in which portion of the pollution gradient the response threshold of a certain biota parameter is located. Secondly, it is not known a priori whether the response thresholds of various biota parameters coincide. In this case measurements of the biota parameters give fundamentally new information compared to the pollution data. Moreover, if the nonlinear model is correct, then biota parameters are estimates of the NEQ as such, while the pollution data are predictors of the NEQ only.

An important difference between the linear and nonlinear models is the accuracy of NEQ measurement. In the case of the linear model we measure the quality in ratio scale (using this scale) such as body mass, many measurements in physics are made using this scale. Usage of such a scale allows exact measurements (the amount of quality gradations is 10^1–10^2). With the nonlinear model we deal with another scale – the ordinal scale, where measurement accuracy is considerably lower (the amount of the quality gradations is 10^0–10^1) and only the succession of gradations may be revealed, not how much the quality has changed. This results from the fact that using the S-curve of the dose-response relationships we can objectively distinguish only three states of a certain biota parameter (normal, buffer, and impact), which arrange the continuum of quality changes into "discrete" intervals, within which quality is homogeneous.

Thus the transition from the presently predominant and more simple (but too simplified) linear model to a more complex (but more precise) nonlinear model leads to a shift of paradigms in environmental quality evaluation.

7. THE STAGES OF ECOSYSTEMS TRANSFORMATION AS A HIGHLY AGGREGATED INDEX

Many authors have concluded the analysis of ecosystem changes in industrialy polluted regions by the description of the stages of ecosystems transformation (see, for example, Lukina and Nikonov 1991; Kryuchkov 1993; Alexeyev; Arzhanova and Elpat'evskii 1990; Bormann 1982; Kerzhentsev 1985). Usually these stages are interpreted as phases of allogenic succession (mostly retrogressions) and can be distinguished based on very different criteria, from the succession of vegetation taxons up to the change of structure and the dynamics of an ecosystem as a whole. Examples of some schemes of transformation suggested by different authors are presented in Figure 32.5(a).

Usually schemes of transformation are considered as a purely subjective category with didactic rather than measurement functions. Two points need to be emphasized here: firstly, that various authors using very different criteria and working in quite different natural conditions, frequently distinguish a similar quantity of stages (4–5); secondly, that boundaries among the stages in the schemes given by various authors significantly coincide. This of course may result exclusively from psychological peculiarities of the researchers, but it more probably has a certain objective basis. The conclusion of this article, concerning nonlinearity of an ecosystem response to load, is that transitions between stable states are "fast", therefore zones of the transition are spatially small and, consequently, the error of diagnostics of any particular stage is not very great. In this sense stages of transformation are objective.

Thus it follows that stages of transformation may be used as a highly aggregated index of NEQ. The stage of transformation is invariant of quality. No matter by which criteria we distinguish the stages, we interpret them in terms of quality – as a normal state or a certain state of pre-pathology, pathology, or death of an ecosystem. It is also important that the index depends only on the NEQ, and is independent from the specific features of particular ecosystems (their type, species composition of the biota, and peculiarities of structure) and the specific features of natural conditions.

The method of index formation is shown in Figure 32.5(b). The index measures NEQ in an ordinal scale, the number of gradations being 3–6 (depending on how much the important or sensitive unimportant parameter states overlap).

The greatest disadvantage of various highly aggregated indices is their artificialness. This in turn causes other shortcomings – nonsensitivity to variously directed changes of indicators, high sensitivity to strong accidental deviations, and the difficulty of interpretation by policy-makers. A stage of transformation is an index aggregated by the nature itself and therefore lacking these disadvantages. The environmental index based on the transformation stage is a very robust statistic: it is dependent purely on errors and fluctuations during the measurement of each

(a)

1	primary phytocoenosis (Piceetum fruticuloso-hylocomiosum)		Piceetum hylo-comioso-fruticulosum	Piceetum graminoso-fruticulosum	Sparse Piceetum empetrosum	industrial barren
2		very initial degradation of ecosystems	initial degradation of ecosystems	severely destructed ecosystems	severely damaged ecosystems	total destruction
3	undamaged forest		initially damaged forest	damaged forest	strongly damaged forest	total destruction
4		moderate impact		strong impact	intensive impact	most intensive impact
5		elimination of sensitive species		change of structure	partly destroyed ecosystem	collapse of ecosystem
6	normal fluctuation	external fluctuation	chronic change	acute reversible change	acute unreversible change	total degradation
7	normal state		pre-pathology	pathology		death

(b)

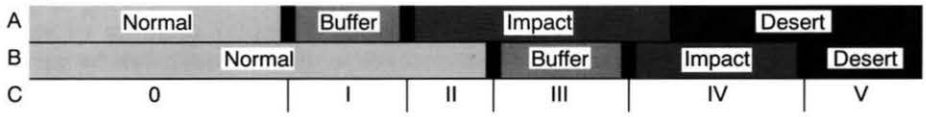

FIGURE 32.5 *Ecosystem transformation stages (a) and construction of an environmental index (b). Criteria for stages distinguishing: 1 – vegetation (Lukina and Nikonov 1991), 2 – vegetation, lichens, and soil (Kryuchkov 1993), 3 – tree health and lichens (Alexeyev), 4 – landscape biogeochemistry (Arzhanova and Elpat'evskii 1990), 5 – ecosystem structure (Bormann 1982), 6 – ecosystem dynamics (Kerzhentsev 1985), 7 – ecosystem quality. A – state of unimportant but sensitive parameters, B – state of important parameters, C – environmental index (number of stage).*

particular parameter. Furthermore, to some extent it depends purely on the choice of parameters to be measured, i.e. the same result will be obtained if different (but essentially overlapping) sets of parameters are registered. In other words, inclusion of every new parameter besides a certain minimal set strengthens the reliability of the results without basically changing them.

In this article the problem of diagnosis of transformation stages has not been covered, as this is a separate field of investigation. The intention has been to pay attention to the possible advantages of using transformation stages as an environmental index.

8. CONCLUSION

Nonlinear effects are well known in ecology. This article has emphasized that nonlinearity of ecosystem response exists not only in theory but in practice too,

and that this is a very important effect to take into account during environmental quality evaluation

Discussions of the problem of environmental indices conventionally consider economic indices (such as the Gross Domestic Product) as examples to imitate. The choice of such examples is based on the tacit assumption that the environmental quality can be measured very exactly. The basic idea of this article is that this is impossible to achieve, not because it is difficult but because of great restrictions in the nonlinear character of ecosystem response. Consequently, attempts at a very precise measurement of quality by constructing various indices will fail: the quality may be measured only roughly and will not deceive policy-makers. Furthermore, rough estimates of the quality (like the suggested index of ecosystem transformation stages) are much more understandable by policy-makers than various artificial indices.

REFERENCES

Alexeyev, V.A. Some problems of diagnostics and classification of forest ecosystem damaged by pollution. In *Forest Ecosystems and Air Pollution*, ed. V.A. Alexeyev, 38–54. Nauka, Leningrad (in Russian).

Alexeyev, A.S. and Tarasov, E.V. 1990. Assessment of relationship between spruce forest state and snow pollution. In *Ecology and Forest Protection*, 3–7. Forestry Engineering Institute, Leningrad (in Russian).

Armand, A.D., Kaidakova, V.V., Kushnareva, G.V., and Dobrodeev, V.G. 1991. Determination of the limits of geosystems' stability on the example of the vicinity of the Monchegorsk metallurgical works. *Izv. Akad. Nauk SSSR, Ser. Geografich.*, 1: 93–104 (in Russian).

Arzhanova, V.S. and Elpat'evskii, P.V. 1990. *Geochemistry of Landscapes and Technogenesis*. Nauka, Moscow (in Russian).

Bormann, F.N. 1982. The effects of air pollution on the New England landscape. *Ambio*, 11: 338–346.

Fedorov, V.D. 1976. The problem of maximum tolerance influences of anthropogenic factor from the ecologist's point of view. In *Comprehensive Analysis of the Environment*, 192–211. Gidrometeoizdat, Leningrad (in Russian).

Freedman, B. and Hutchinson, T.C. 1980. Long-term effects of smelter pollution at Sudbury, Ontario on forest community composition. *Can. J. Bot.*, 58: 2123–2140.

Kaigorodova, S.Yu. and Vorobeichik, E.L. 1996. Changes in certain properties of grey forest soil polluted with emission from copper-smelting plant. *Russian J. of Ecology*, 27: 177–183.

Kerzhentsev, A.S. 1985. Preface. In *Air Pollution and Forests*, ed. W.H. Smith, (Russian translation), 5–21. Progress, Moscow (in Russian).

Kryuchkov, V.V. 1993. Degradation of ecosystems around the "Severonikel" smelter complex, 35–46. *Proceedings of the International Workshop Aerial Pollution in Kola Peninsula*, St. Petersburg, Kola Scientific Centre, Apatity.

Lukina, N.V. and Nikonov, V.V. 1991. Changes in the primary yield of spruce stands as affected by air-borne industrial pollution in the Kola Peninsula. *Lesovedenie*, 4: 37–45 (in Russian).

Mikhailova, I.N. and Vorobeichik, E.L. 1995. Epiphytic lichenosynusia under conditions of chemical pollution: dose-effect dependencies. *Russian J. of Ecology*, 26: 425–431.

Moiseenko, T.I. and Kudrjavceva, L.P. 1995. Nickel in Kola Peninsula surface water, its accumulation and toxic effects. In *Problems of Chemical and Biological Monitoring of Kola North Water Ecosystem*, 36–45. Apatity.

Nordgren, A., Baath, E., and Söderström, B. 1985. Soil microfungi in an area polluted by heavy metalls. *Can. J. Bot.*, 63: 448–455.

Saliev, A.V. 1988. Modelling of the action of atmospheric phytotoxic substance on plants – spatial aspect. In *Principles of Biological Monitoring of Environmental Pollution*, 137–160. Institute of Applied Geophysics, Moscow (in Russian).

Stepanov, A.M. (ed.) 1992. *Comprehensive Ecological Evaluation of Technogenic Action on Ecosystem of the Southern Taiga*, Centre of Ecology and Productivity of Forests (CEPF). Moscow (in Russian).

Stepanov, A.M. 1995. Atmospheric pollution limits. In *Forest Ecosystem of Kola Peninsula Under Atmospheric Pollution Influence of Smelters*, ed. V.V. Sychev, 176–210, Centre of Ecology and Productivity of Forests (CEPF). St. Petersburg (in Russian).

Trubina, M.R. and Mahnev, A.K. 1997. Dynamic of the forest ground vegetation under condition of chronic fluorine pollution. *Russian J. of Ecology*, 28: 90–95.

Tsvetkov, V.F. 1993. Threshold levels of air pollution and forest decline in surroundings of "Severonikel" smelter complex, 397–401. *Proceedings of the International Workshop Aerial Pollution in Kola Peninsula*, St.Petersburg, Kola Scientific Centre, Apatity.

Tyler, G. 1974. Heavy metal pollution and soil enzymatic activity. *Plant & Soil*, 41: 303–311.

Vorobeichik, E.L. 1995. Changes in thickness of forest litter under chemical pollution. *Russian J. of Ecology*, 26: 252–258.

Vorobeichik, E.L. 1998. Populations of earthworms (Lumbricidiae) in forests of the Middle Urals in conditions of pollution by discharge from copper works. *Russian J. of Ecology*, 29: 85–91.

Vorobeichik, E.L. and Khantemirova, E.V. 1994. Reaction of forest phytocenoses to technogenic pollution: dose-effect dependencies. *Russian J. of Ecology*, 25: 171–180.

Vorobeichik, E.L., Sadykov, O.F., and Farafontov, M.G. 1994. *Ecological Standardization of Terrestrial Ecosystem Technogenic Pollution (local scale)*. Nauka, Ekaterinburg (in Russian).

Yeates, G.W., Orchard, V.A., and Speir, T.W. 1995. Reduction in faunal populations and decomposition following pasture contamination by Cu-Cr-As based timber preservative. *Acta Zool. Fen.*, 196: 297–300.

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