

Chapter 9

Effects of Industrial Polluters: General Patterns and Sources of Variation

9.1 The State of Knowledge

We identified about 2,500 data sources that report impacts of point polluters on terrestrial biota and fit criteria listed in Section 5.1.2. We think that our collection is sufficiently representative and covers at least a half of existing publications. Although we have started a systematic review of these publications, we estimate that the completed meta-analyses (Ruotsalainen & Kozlov 2006; Zvereva et al. 2008; Zvereva & Kozlov 2009; Roitto et al. 2009) still cover less than 25% of the available information. The aim of Section 9.1 is to briefly evaluate knowledge of changes of landscapes, ecosystems, communities, populations and individual organisms in impact zones of industrial enterprises in order to reveal critical shortcomings and data gaps.

9.1.1 Landscape-Level Effects

The emissions of point polluters can act in such a way that the consequences of pollution-induced changes become the dominant feature of the landscape (Bohne 1971; Broto et al. 2007). However, integrated studies of polluted landscapes are infrequent (Gilbert 1975; Doncheva 1979; Kozlov & Zvereva 2007a), and they only rarely include the societal dimension (Broto et al. 2007). Investigation of pollution impact on biota requires a reduction of scale that may not be appropriate for the study of landscapes; therefore, landscape-level effects were not explored in the course of our project. However, these effects are of critical importance for understanding the indirect impacts of industrial pollution on biota, and we would like to stress the need for exploration of specific landscapes that have evolved around point polluters.

Landscape-level effects are easily recognisable around many of polluters considered in this book. Big polluters dominate both land and sky, especially when puffs of smoke make them visible from large distances (please see color plates 18, 50, 61, 64 and 65 in Appendix II), and are often surrounded by extensive areas of disturbed

land (please see color plates 20–23, 32–38, 45–47, 53 and 54 in Appendix II). The visual perception of this disturbance is, in the first line, linked with vegetation damage. Forest death (please see color plates 32 and 33 in Appendix II) is often seen as the most important change, possibly also due to the symbolic connotations of the forest. This phenomenon attracted emotional public attention: ‘Bushes shrank and vanished. Grasses died away. Blighted land replaced the forest. All around us dead hills, red, raw, ribbed by erosion, stood stark in the sunshine. Hardly two miles from dense woodland, we were in the midst of a moonscape on earth.... We were in the southeast corner of Tennessee, in the Ducktown Desert of the Copper Basin’ (Teale 1951). On the other hand, the visual appearance of vegetation in heavily polluted treeless habitats (e.g., tundra) does not differ from unpolluted sites (please compare color plates 66 and 67 in Appendix II) and therefore causes less public awareness. Last but not least, low levels of pollution may cause no visible changes (please compare color plates 56 and 57 in Appendix II), or slight effects of pollution may be masked by consequences of other, stronger impacts associated with human settlements, such as cutting of timber, littering, and recreational activities (please compare color plates 2 and 3, 16 and 17, 40 and 41 in Appendix II).

Extreme depositions of airborne pollutants, especially of sulphur dioxide accompanied by heavy metals, resulted in the appearance of industrial barrens - bleak open landscapes, with only small patches of vegetation surrounded by bare land (please see color plates 9, 21, 29, 34–36, 54 and 59 in Appendix II). These landscapes appeared as a by-product of human activities about a century ago and were first reported in the North America. Recently, we collected information on about 40 industrial barrens worldwide (Kozlov & Zvereva 2007a, and unpublished). Among polluters, the effects of which are documented in the present book, industrial barrens occur around non-ferrous smelters at Harjavalta, Karabash, Krompachy, Monchegorsk, Norilsk, Nickel, Revda, and Sudbury. The extent of industrial barrens varies from few hectares (Harjavalta, Krompachy) to several hundreds of square kilometres (Norilsk, Monchegorsk). In spite of a general reduction in biodiversity, industrial barrens still support a variety of life, including regionally rare and endangered species, as well as populations that evolved specific adaptations to the harsh and toxic environments (Kozlov & Zvereva 2007a).

Pollution often changes the visual perception of landscapes, making them unattractive - especially when dead trees, which remain standing for decades (please see color plates 32, 33, 47 and 49 in Appendix II), cut the skyline (Gilbert 1975). However, standing trees, even when they are dead, maintain climatic and biotic stability of contaminated habitats, in particular by ameliorating microclimate (Wolk 1977) and preventing soil erosion. The importance of standing trees was proven experimentally: the introduction of dead birches to tundra created favourable microclimate for field layer vegetation in spring and early summer (Molau 2003). Cutting of dead trees has been performed in many severely polluted areas (Gilbert 1975; Kozlov & Zvereva 2007a); this action improved visual attractiveness of landscapes but accelerated secondary damage. The old clearcuts under severe pollution impact near both Monchegorsk and Nickel have been rapidly transformed to industrial barrens, while in the adjacent uncut areas some vegetation is still alive (Kozlov 2004).

9.1.2 Ecosystem-Level Effects

Ecosystem-level processes form a focal point in the development of the theoretical basis of both ‘basic’ and ‘applied’ ecology. Appreciation of the value of ecosystem services for human well-being (Costanza et al. 1997) further stresses the need to explore basic principles governing ecosystem development in a changing world (Kremen & Ostfeld 2005; Mokany et al. 2006; Grimm et al. 2008). However, exploration of ecosystem-level effects is a difficult task, and therefore some predictions of ecosystem changes due pollution impacts (Newman & Schreiber 1984; Odum 1985; Rapport et al. 1985) remain too general to derive testable hypotheses.

A decrease in net primary productivity is considered one of most important ecosystem-level responses to different kinds of disturbances (Odum 1985; Rapport et al. 1985; Sigal & Suter 1987; Treshow & Anderson 1989; Barker & Tingey 1992; Armentano & Bennett 1992; Newman & Schreiber 1984; Bezel 2006). Since biomass or standing crop is a primary variable measured for determining productivity (Sigal & Suter 1987), pollution effects on productivity can be deduced from overall decreases in stand basal area and stand height, as well as from decreases in cover of field layer vegetation and mosses found in our study (Chapter 6). Combined with reduction of shoot length, radial increment, and root growth (Roitto et al. 2009; Chapter 4), these data suggest that a decrease in net primary productivity with pollution is likely to be a general ecosystem-level phenomenon.

Decreased productivity is a pre-requisite of changes in nutrient cycling. Decreases in both basal area and stand height as well as in cover of field layer vegetation (Chapter 6) are indicative of lower litter input and, consequently, lower forest floor thickness (Keane 2008; Merino et al. 2008). A decrease in topsoil depth around non-ferrous smelters (Chapter 3) is consistent with this conclusion. On the other hand, pollution adversely affects soil microbiota (Bååth 1989; Fritze 1992; Ruotsalainen & Kozlov 2006; Chapter 3) and detritophagous arthropods (Zvereva & Kozlov 2009), substantially decreasing the biological decomposition rate and breakdown of the litter (Rühling & Tyler 1973; Prescott & Parkinson 1985; Zwolinski 1994; McEnroe & Helmisaari 2001).

The decreases in both production and decomposition of organic matter are so far the only ecosystem-level responses to pollution, the generality of which is supported by the available information. These effects are in line with the prediction (Odum 1985) that polluted ecosystems become more open: reduction of internal nutrient cycling increases the importance of both input and output environments.

9.1.3 Community-Level Effects

An excellent book by Clements and Newman (2002) provides a broad perspective for bridging ecological and toxicological approaches in the study of community-level effects of pollution on biota. One of the most important conclusions made by

these authors concerns the need to pay more attention to the indirect effects of contaminants, mediated by changes in species interactions, such as competition, facilitation, predation and mutualism.

Species diversity is the most frequently measured community parameter (Sigal & Suter 1987). However, the results of case studies are often over-generalized, and therefore they are discussed in Section 9.2.2. Insufficient knowledge of pollution impacts on many groups of terrestrial biota further restricts discussion to changes in vegetation structure and in arthropod communities.

Armentano and Bennett (1992) listed the following responses of plant communities to chronic air pollution: reduced photosynthesis, reduced labile carbohydrate pool, reduced growth of root tips and new leaves, decreased leaf area, differences in species growth performance, reduced community canopy cover, reduced reproductive capacity, shifts in interspecific competitive advantage, alteration of community composition, change in species diversity, and change in community structure (physiognomy). However, our data provided no support for most of these predictions. In particular, overall effect of pollution on leaf/needle size and shoot length of woody plants did not differ from zero (Figs. 4.8 and 4.14). Individual species of woody plants similarly responded to pollution in terms of both leaf/needle size and shoot growth (Figs. 4.10 and 4.16). Canopy cover also did not change with pollution (Fig. 6.1), and no changes in either species richness of vascular plants (Fig. 6.9) or physiognomy of vegetation were observed around many of the studied pollutants (please compare color plates 2 and 3, 16 and 17, 40 and 41, 56 and 57, 66 and 67 in Appendix II).

The differential responses of some taxonomic and functional groups of both plants and arthropods to pollution (Zvereva et al. 2008; Zvereva & Kozlov 2009; Chapters 4–5) support the opinion that industrial pollutants affect the trophic structure of ecosystems (Sigal & Suter 1987). In particular, pollution generally imposes larger detrimental effects on predators than on herbivorous arthropods, thus creating so-called enemy-free spaces for herbivores. This phenomenon was documented by several case studies (Zvereva & Kozlov 2000a, b) and confirmed by meta-analysis of published data (Zvereva & Kozlov 2009). At the same time, the available information remains insufficient to support the hypothesis on simplification of ecosystem structure (Odum 1985; Rapport et al. 1985; Freedman 1989; Armentano & Bennett 1992) or on the ‘loss of ecosystem integrity’ (Barker & Tingey 1992) under pollution impact.

Another frequently repeated statement (Newman & Schreiber 1984; Bezel 2006) is an overall decrease in the size of organisms with pollution. This hypothesis was suggested by Odum (1985), who referred to mesocosm experiments with plankton (that were presumably unpublished at that time), and by Rapport et al. (1985), who based his generalizations on observational studies of vegetation decline due to pollution (Gordon & Gorham 1963) and irradiation (Woodwell 1967). Surprisingly, we found almost no attempts to verify this hypothesis (but see Braun et al. 2004), and therefore its validity and generality remain unknown.

Thus, although some responses to industrial pollutants are common for the communities explored so far, natural variation, rapid development of adaptations and

intrinsic community differences complicate any generalization. Importantly, our inability to reveal consistent and significant effect of pollution on most of the explored community characteristics results not from the absence of effects, but from the diversity in responses often yielding a zero overall effect.

9.1.4 Population-Level Effects

Changes in both population structure and some attributes of population dynamics are reported for a number of species. However, only a few species were explored in sufficient detail, and only a limited number of parameters were studied in more than a few species. This naturally limits our understanding of changes in population behaviour under pollution impacts.

Pollution influences the genetic structure of populations, often leading to the development of pollution tolerance. These effects, referred as micro-evolution due to pollution (Medina et al. 2007), are best documented in microbiota; changes in the genetic structure of plant populations were reported more frequently than in animal (mostly arthropod) populations. The heavy metal tolerance of plants, especially grasses, has been studied extensively and provides a well-documented example of rapid evolutionary adaptation (Bradshaw & McNeilly 1981; Shaw 1990; Macnair 1997). Even long-lived trees may develop pollution resistance to both sulphur dioxide and heavy metals (Rachwal & Wit-Rzepka 1989; Turner 1994; Utriainen et al. 1997; Eränen 2008), presumably through survival selection (Kozlov 2005b).

Micro-evolution due to pollution is likely to be more common than has been documented (Barker & Tingey 1992; Kozlov & Zvereva 2007b; Eränen 2008). Therefore, a currently identified (Medina et al. 2007) gap between studies addressing changes in the genetic structure of populations and those assessing effects at higher levels of biological organisation should be overcome to improve understanding of ecosystem responses to pollution. Bridging molecular and community-level studies may revitalise exploration of phenotypic differentiation of populations living in contaminated environments (Kozlov 1990; Newman et al. 1992; Zvereva et al. 2002).

Surprisingly, we have not found any study documenting changes in the spatial structure of plant populations. For animals, fragmentary data exist on insect herbivores (Kozlov 1990, 2003) and small mammals (Mukhacheva 2007). The only conclusion that can be made at the moment is that pollution may change small-scale distribution patterns of organisms, although the mechanisms behind these changes remain nearly unknown.

Studies addressing the pollution impact on population demography conclude that contaminated populations usually differ from control populations in age structure (Zverev et al. 2008). The impacted populations may be both younger (Kucken et al. 1994; Zhuikova et al. 2001) and older (Deyeva & Maznaya 1993; Dmowski et al. 1998; Zverev et al. 2008; Zverev 2009) than the control ones. Since morphological and physiological parameters of plants and animals generally change with age, some of the differences between populations from contaminated and control sites result

from changes in age structure rather than from direct effects of pollutants on individual fitness (Zverev et al. 2008).

Even less is known on sex ratio in the affected populations. Although the medical literature suggests that industrial pollution may change sex ratios (Williams et al. 1992; Lloyd et al. 1984, 1985), some studies found no convincing evidence of this effect (Williams et al. 1995; Kozlov 1999). Isolated observations on animal populations in contaminated areas also produced diverse results. The proportion of females of two insect species increased in surroundings of chemical factories (Birg 1989; Chumakov 1989). Male local survival of the pied flycatcher (*Ficedula hypoleuca*) was higher near the non-ferrous smelters at Harjavalta relative to unpolluted habitats (Eeva et al. 2006b), potentially resulting in sex ratio changes along pollution gradients. Finally, no links between pollution load and sex ratio was found in populations of two vole species (*Clethrionomys rufocanus* and *C. glareolus*) near non-ferrous smelters located at Monchegorsk and Revda, respectively (Kataev 1984; Vorobeichik et al. 1994). Variation in sex ratio in soil nematode communities around the Almalyk industrial complex, reported by Pen-Mouratov et al. (2008), was independent of copper concentration in soils of the study sites.

Although abundance is one of the most frequently measured population characteristics, pollution effects on long-term density fluctuations have only rarely been studied (Zvereva et al. 1997b, 2003; Kozlov 2003; Zvereva & Kozlov 2004). First and most importantly, the majority of research projects last 1 to 3 years, thus yielding estimates of population density rather than monitoring population dynamics. So far, only a few species or species groups have been monitored along a pollution gradient uninterruptedly for 10 or more years. Furthermore, pollution effects on a suite of population parameters driving population dynamics, such as death rate, life expectancy, survival and fecundity, are properly documented for a very few species of terrestrial biota.

Especially surprising is absence of information on wildlife mortality in pollution gradients, possibly because direct measurements of death rates seem nearly impossible. Although observations on mortality of cattle and wildlife attributable to air pollution have repeatedly been summarised (Newman 1979; Newman et al. 1992), they almost exclusively concern extreme levels of exposure or catastrophic local events. So far, only indirect data, derived from measurements of abundance of necrophagous beetles, suggest that mortality of vertebrates increases near point polluters (Kozlov et al. 2005a). It seems that modelling of pollution effects on population dynamics (Kidd 1991; Dubey 2004, and references therein) is developed much better than validation of these models with observational data.

Review of available knowledge allowed us to identify a few species for which the amount of available information is sufficient or nearly sufficient to approach understanding of pollution effects on population dynamics. The list includes white birch, *Betula pubescens*; Scots pine, *Pinus sylvestris*; bilberry, *Vaccinium myrtillus*; willow feeding leaf beetle, *Chrysomela lapponica*; Great tit, *Parus major*; pied flycatcher, *Ficedula hypoleuca*; grey-sided vole, *Clethrionomys rufocanus*; and bank vole, *C. glareolus*.

Importantly, comparison of population abundances between polluted and unpolluted study sites can provide little information about pollution effects on population dynamics. The same species can demonstrate different patterns of density changes not only around different polluters, but also in different years around the same polluter. For example, the population density of the leaf beetle, *Chrysomela lapponica*, around a nickel-copper smelter at Monchegorsk demonstrated a significant dome-shaped pattern in 1994 (Zvereva et al. 1995), but not in 1998, when it gradually decreased from the most to the least most polluted sites (Zvereva et al. 2002) nor in 2006–2007 when the density of this species did not change with distance from the polluter (E.L.Z. & M.V.K., personal observation, 2007). Similar variation in density patterns along the Monchegorsk pollution gradient was observed in *Eriocrania* leafminers during 1994–2005 (Zvereva & Kozlov 2006b). Several other studies reporting multiyear density data (Selikhovkin 1995, 1996; Kozlov 2003) also suggest that all kinds of pollution-effect relationships can be observed for the same species during different study years, as a result of asynchronous density fluctuations in spatially isolated populations (Selikhovkin 1995; Zvereva et al. 2002; Kozlov 2003; Zvereva & Kozlov 2006b). Therefore, classification of species on the basis of density changes along pollution gradients, widely accepted in narrative reviews on insect responses to pollution (Alstad et al. 1982; Führer 1985; Kozlov 1990; Heliövaara & Väisänen 1993), is hardly reasonable. However, this does not mean that further accumulation of short-term data on population abundances along pollution gradients is useless. Although short-term observations are of limited value for understanding pollution impacts on the population dynamics of individual species, meta-analysis of these observations can yield important conclusions on overall effects of pollution on population densities of different taxonomic or functional groups of biota.

9.1.5 Organism-Level Effects

A substantial number of observational studies conducted around point polluters focused on responses of individual organisms. Diversity of both organisms and approaches hampers generalization of these data. However, the critical problem, identified two decades ago (Sigal & Suter 1987), is a common absence of linkages between the recorded individual responses and processes occurring in the affected populations and communities.

A number of publications reported increases in the visible injury of plants with pollution (Jacobson & Hill 1970; Sigal & Suter 1987; Treshow & Anderson 1989; Flagler 1998). However, this field of knowledge, the oldest one in the scientific documentation of the effects caused by pollution, has mostly developed in isolation from studies addressing changes in plant growth and reproduction. A variety of symptoms, subjectivity in estimation of damage, common absence

of statistical analysis, especially in older publications, and absence of links between visible injury and changes in fitness made interpretation of the accumulated data rather vague (Percy & Ferretti 2004). This also concerns various indices of tree health used mostly by Russian scientists (Alexeyev 1995) and visual estimates of crown defoliation and discoloration, serving as the basis of large-scale forest monitoring program (UN-ECE 2006). Although all these indices are potentially useful for monitoring purposes, we need to stress the low value of publications based on various estimations of visible injury for the development of pollution ecology.

Body size or weight is the only organism-level characteristic that was reported in a sufficient number of studies. Meta-analysis of data on terrestrial arthropods demonstrated that body size significantly decreased with pollution (Zvereva & Kozlov 2009). Observations on birds and mammals (Bezel 2006; Veličković 2007a), although not yet summarised, seem to show a similar trend. Thus, a decrease in body size is likely to be a general regularity for animals.

9.2 Myths of Pollution Ecology

Profound concern for environmental quality, amplified by media attention, resulted in both diminution and exaggeration of particular problems by both representatives of industries and environmental activists. These processes not only biased the ideas of the general public on effects caused by pollution, but also unavoidably influenced scientists. Prejudgement and misinterpretation presumably imposed even stronger adverse impacts on the development of pollution ecology than the methodological shortcomings discussed in Section 8.4. In short, there is no doubt that industrial pollution was the primary cause of extreme environmental deterioration in many areas (Gordon & Gorham 1963; Wood & Nash 1976; Freedman & Hutchinson 1980b; Kozlov & Zvereva 2007a), but these severe effects are neither general nor widespread. Only some authors (Barker & Tingey 1992; Clements & Newman 2002) appreciate unequivocally that the consequences of air pollution to biota and the resulting impact on ecosystem structure and functions (services) are not clearly known. We strongly support this opinion.

9.2.1 *Generality of Responses*

It has been repeatedly claimed that there are commonly observed patterns of ecosystem responses to stress (Woodwell 1970; Lugo 1978; Auerbach 1981; Odum 1985; Rapport et al. 1985; Freedman 1989), while variation in responses is rarely appreciated (Rapport & Whitford 1999; Clements & Newman 2002). This approach results in an exaggerated impression of both the generality and the uniformity of the effects caused by pollution.

A random sample of 100 publications from our data base (including 50 papers in Russian and 50 papers by native English speakers) demonstrated that the investigated point polluter was explicitly named (or associated with a certain town) in the titles of only ten publications. Titles of another 16 publications, although not pointing out at the polluter, identified the geographical region where the impact zone was located. Thus, titles of three quarters of publications reported just 'pollution impact on...' or something similar, unconsciously assuming that the effects are general and uniform across both polluters and impacted (stressed) communities.

'Stress' is used loosely by ecologists to describe the relative (deviating from the presumed optimum) circumstances affecting species and communities (Lortie & Callaway 2006). As a result, 'low-stress' and 'high-stress' environments are often labelled arbitrarily, on the basis of the researcher's assumptions about which conditions are favourable or unfavourable for the organisms or communities under study. However, whether these organisms or communities really experience stress in 'high-stress' environments is only rarely questioned. Instead, any differences between 'stressful' and 'benign' environments are routinely labelled as 'stress responses'. Generalization of these 'stress responses' is likely to produce misleading results, especially when the differences between the 'low-stress' and 'high-stress' environments are not accounted for (Lortie & Callaway 2006).

Grime (1979) defined stress in terms of productivity. Accordingly, stressful environments are defined as those in which producers are limited in their ability to convert energy to biomass. This approach is commonly used by ecologists to examine stress across communities (Underwood 1989; Goldberg et al. 1999; Parker et al. 1999; Kammer & Mohl 2002; Lortie & Callaway 2006). Thus, an overall decrease in productivity (Section 9.1.2) can be seen as a proof of ecosystem-level stress caused by industrial pollution. However, this does not necessarily mean that all organisms and populations experience stress under pollution impact. What is stressful for one species may be optimal for another species, or even for another population of the same species, because of differences in adaptation to particular environments (Lortie et al. 2004).

Pollution can sometimes even improve environmental conditions for certain species relative to the benign (unpolluted) habitats. For example, the absence of competitors resulted in faster growth of birch seedlings on metal-contaminated soils relative to unpolluted forests with a dense cover of field layer vegetation (Eränen & Kozlov 2009). Similarly, decreased pressure from natural enemies in polluted habitats and sometimes also improvements in host plant quality favoured density increase of some herbivorous insects, in spite of adverse effects of pollutants on the individual performance of these insects (Kozlov et al. 1996b; Zvereva & Kozlov 2000a, b, 2009).

We conclude that a widespread opinion on the existence of a general pattern of ecosystem responses to pollution results mostly from earlier generalizations based on the few most impressive case studies that were available that time (Haywood 1907; National Research Council of Canada 1939; Gorham & Gordon 1960a, b; Gordon & Gorham 1963; Hutchinson & Whitby 1974; Wood & Nash 1976; Freedman & Hutchinson 1980a, b; Amiro & Courtin 1981). We suggest that efforts

should be directed primarily at exploring sources of variation in responses to pollution, rather than at searching for the presumed general effects.

9.2.2 *Uniformity of Responses*

The majority of the existing reviews claim that pollution effects on organisms, populations, communities and ecosystems are generally detrimental (Alstad et al. 1982; Newman & Schreiber 1984; Sigal & Suter 1987; Bååth 1989; Riemer & Whittaker 1989; Treshow & Anderson 1989; Barker & Tingey 1992; Fritze 1992; Heliövaara & Väisänen 1993; Rusek & Marshall 2000; Bezel 2006). Even if some organisms benefit from pollution, this effect is usually interpreted as the sign of an overall decrease in ecosystem vitality or integrity (e.g., pests benefit from plant weakening by pollution and accelerate destruction of plant communities by imposing additional damage: Wentzel & Ohnesorge 1961; Carlson et al. 1977; Baltensweiler 1985; Führer 1985; Berger & Katzensteiner 1994). Although in many situations environmental pollution obviously harms organisms, communities, and ecosystems, this blackening perception is not always true. The adaptation potential of individual organisms and entire ecosystems should not be underestimated.

Decreases in net primary productivity (discussed in Section 9.1.2) and in diversity are most frequently mentioned among the general effects of pollution on ecosystem properties (Odum 1985; Rapport et al. 1985; Sigal & Suter 1987; Treshow & Anderson 1989; Barker & Tingey 1992; Armentano & Bennett 1992; Newman & Schreiber 1984; Bezel 2006). However, we are not aware of any study documenting ecosystem-level diversity along pollution gradients.

The pollution effects on diversity of different groups of biota are not uniform. While the diversity of soil microfungi (Ruotsalainen & Kozlov 2006), vascular plants (Zvereva et al. 2008; Chapter 6), and soil arthropods (Zvereva & Kozlov 2009) generally decreased with pollution, the species richness of insect herbivores increased (Zvereva & Kozlov 2009). Moreover, the reported adverse effects on diversity may appear overestimated, since the researchers generally did not account for pollution effects on species abundance (Zvereva & Kozlov 2009). As we demonstrated earlier, the confounding effect of density may change even the sign of the pollution effect on diversity (Kozlov 1997). Thus, both methodological problems and individualistic responses of different groups of biota make generalization of pollution effects on ecosystem-level diversity premature.

We discovered the diversity in responses of the studied components of terrestrial ecosystems by meta-analyses of both published (Ruotsalainen & Kozlov 2006; Zvereva et al. 2008; Zvereva & Kozlov 2009; Roitto et al. 2009) and original data (Chapters 3–7). We consider this discovery as one of the most interesting and important findings of our research. Identification of factors explaining this variation (partially discussed in Section 9.4) is crucial for understanding and predicting pollution-induced changes in affected ecosystems.

9.2.3 *Causality of Responses*

It seems that ‘presumption of guilt’ is commonly accepted by pollution ecologists, and the majority of changes observed around industrial polluters is unthinkingly attributed to the toxicity of pollutants. This is especially common for adverse effects, like the death of cattle, crop damage, or forest decline. Causality is discussed only rarely (Courtin 1994), mostly in situations when effects are somewhat unexpected, such as better plant growth near the polluter (Zvereva et al. 1997a), or absence of the pollutant at the locality when damage occurs (Tikkanen & Niemelä 1995).

Reviews and methodological papers addressing pollution impact on biota usually do not even mention the need to carefully explore the causes of the phenomena observed in polluted environments (Schindler 1987; Sigal & Suter 1987; Freedman 1989; Treshow & Anderson 1989; Barker & Tingey 1992; Bezel 2006) or briefly advertise experimental studies as a tool to demonstrate cause-and-effect relationships (Clements & Newman 2002). However, although the randomised experiment is often invoked as one method for unambiguously determining causality (Clements 2004), even that is questionable when outcomes are stochastic and we rely on statistical interpretation (Fabricius & De’ath 2004).

Pollution ecologists are not the first scientists to face the attribution of causality problem. The concept of causality has a long and complex history. Its everyday usage is often straightforward, but as a philosophical or scientific concept, its definition and use are often contentious (for more details and references consult Fabricius & De’ath 2004). One of the best known examples of rules for attributing the effect (disease) to its presumed cause (microbe) are the Koch’s postulates (Koch 1880). A much closer example concerns epidemiology, which deals with issues of complexity comparable to the ecological and environmental sciences and has similar requirements of scientific rigour and a limited ability to use the experimental approach.

Epidemiologists have long ago developed criteria to pass from statistically confirmed association ‘to a verdict of causation’ (Hill 1965). These criteria include: strength and consistency of observed association, its specificity and temporality (a logical time sequence of events), existence of the relationship between the dose (the putative cause) and the response, coherence with known biological facts, plausibility, experimental support and analogy with similar, better known, situations (Hill 1965; Susser 1986; Fabricius & De’ath 2004). These criteria, with few modifications, were also suggested for studies addressing effects of pollutants on biota (Fox 1991). Importantly, none of the criteria are taken as indicative by themselves, but equally, none are seen as absolutely necessary to evaluate the causal significance of associations (Roth et al. 1982). The more criteria that are satisfied and the stronger the association, the more confidence we should have in our judgement that the association is causal (Fabricius & De’ath 2004).

While the use of these criteria in an individual case study may be difficult or even impossible (but see Schroeter et al. 1993), their applicability to generalized results of multiple studies casts no doubt. Formalising the conclusions of our review on industrial barrens (Kozlov & Zvereva 2007a), we attribute the development of

these specific landscapes to the effects of aerial emissions of big non-ferrous smelters because:

1. Most of known industrial barrens (33 of 35) are associated with non-ferrous smelters (consistence and specificity criteria).
2. In all documented situations, industrial barrens developed after the beginning of smelting (temporality criterion).
3. The size of the barren area is roughly proportional to the amount of aerial emissions, and its extent is usually consistent with the extent of territory with excessive soil contamination by toxic metals (dose dependence criterion).
4. Adverse effects of sulphur dioxide and toxic metals on biota were demonstrated in a number of experimental studies (coherence criterion).
5. The phenomenological model describing the development of industrial barrens does not contradict any of basic regularities commonly accepted by ecologists (plausibility criterion).

However, while most barrens are associated with non-ferrous smelters, not every smelter is surrounded by industrial barrens. Thus, we need to restrict the scope of our conclusions, but a shortage of information prevents verification of the hypotheses (Kozlov & Zvereva 2007a) that the detected causal link is valid only for mountainous or hilly landscapes and/or regions with relatively harsh climate.

Meta-analysis was recently suggested as a tool to evaluate whether association between an increase in mortality and air pollution is strong enough to be interpreted as evidence for causality (Bellini et al. 2007). However, to our knowledge, there are no guidelines to judge the strength of evidence by the size of the observed effect, and thus the interpretation of the results remains subjective. In contrast, the significance of the overall effect can be seen as the proof of consistency of the observed association.

An overall effect was significant in nine of 25 meta-analyses of original data collected during the course of our project (Figs. 3.1–3.5, 4.1, 4.4, 4.8, 4.11, 4.14, 4.17, 4.19, 4.20, 5.5, 6.1–6.9, 7.1, 7.4). This can indeed be seen as a weak proof of causal association between pollution load and the changes of measured characteristics of terrestrial ecosystems. On the other hand, zero overall effect can signal that our meta-analyses are combining diverse, sometimes even opposite, effects. The search for sources of variation demonstrated significant effects of pollution in at least one subset of our data classified according to three categorical variables (classes of polluters, their geographical position and effects on soil pH) in 12 of 16 meta-analyses with zero overall effect. The heterogeneity was mostly due to differences between classes of polluters and, consequently, between groups of polluters causing soil acidification and alkalisation. Thus, although association between the detected changes in terrestrial biota and ‘pollution in general’ was weak, the strength of association increased when we restrict the search of causal relationships to either non-ferrous smelters or aluminium plants.

Exploration of causality is often complicated by indirect (Section 8.2) or secondary effects of pollutants. Pollution frequently triggers secondary effects that may enhance the primary disturbance in a positive feedback fashion (Courtin 1994; Kozlov & Zvereva 2007a). For example, forest thinning results in increased injury from gaseous pollutants (Norokorpi & Frank 1993) and in higher wind speed (Kozlov 2002), which increases climatic stress both directly and by changes in

snowpack structure and also imposes mechanical damage to plants (Kozlov 2001b). A thin and compact snow layer explains lower soil temperatures recorded in industrial barrens during the wintertime (Kozlov & Haukioja 1997). Changes in microclimate in combination with a pollution-induced decrease in the cold hardiness of conifers (Sutinen et al. 1996) increase the probability of death of extant trees from freezing injury. Accumulation of woody debris, in combination with proximity to large human settlements, increases the risk of occasional fires. The disappearance of trees increases albedo, and thus lowers April-June temperatures, creating less favourable conditions for plant growth (Molau 2003) and causing delays in phenology (Kozlov et al. 2007). In subarctic regions, changes in microclimate following deforestation are often so strong that they may hamper recovery of vegetation even in absence of additional stressors (Arseneault & Payette 1997; Vajda & Venäläinen 2005). Forest decline reduces recovery of topsoil (Chapter 3) and facilitates soil erosion (please see color plates 20, 29, 34, 46 and 54 in Appendix II), with adverse consequences for all components of biota.

The effects mentioned above are apparent only at the landscape scale, and therefore their importance is often underestimated: all changes observed around point polluters are routinely attributed to toxicity of pollutants, while many of them may be caused by habitat changes or by altered microclimate. For example, the relative abundance of lizard species in areas affected by sand-mining and atmospheric fluoride is caused by habitat changes in polluted landscapes: forest species become less common and lizard species from open areas become more common (Taylor & Fox 2001). Similarly, the altered growth form of trees in industrial barrens (please see color plates 9, 10, 29, 34, 39 and 46 in Appendix II) results from combination of increased light availability and strong wind impact rather than from the toxicity of pollutants (Kozlov 2001b; Kozlov & Zvereva 2007a). The importance of secondary effects is confirmed by the failure of small-scale experiments with industrial pollutants (Nygaard & Abrahamsen 1991; Shevtsova & Neuvonen 1997; Zobel et al. 1999) to mimic some of environmental changes observed around the point polluters, such as a decline of field layer vegetation.

We conclude that pollution ecology suffers from generally poor understanding of cause-and-effect relationships. In many situations, the causal links behind the changes observed around point polluters are far from being transparent due to both the natural variation in study systems (Section 8.4.2) and the importance of indirect effects and secondary impacts acting at different scales. Causality of the effects routinely attributed to pollution need to be explored in greater details.

9.2.4 Linearity of Responses

As previously mentioned (Section 8.4.3), correlation and regression analyses are most frequently used to explore relationships between pollution load and characteristics of terrestrial biota. The wide use of linear statistical models indicates that researchers generally presume that the associations under study are linear or at least monotonic.

There is no theoretical background behind the commonly presumed linearity of biotic responses to pollution. Instead, the experience of toxicology clearly demonstrated that when both small (no effect) and extreme (lethal) concentrations are considered, then the dose–effect relationships are usually approximated by the logistic (S-shaped) function (Streibig et al. 1993). Seven response patterns of species diversity indices along the disturbance gradient (three linear and four non-linear) were identified in experiments with herbaceous vegetation (Li et al. 2004). There are sufficient reasons to believe that the logistic function generally describes association between pollution load and characteristics of biota better than the linear model (Vorobeichik et al. 1994). However, a shortage of information prevents verification of this hypothesis: only a thin fraction of studies conducted around point polluters is based on sufficiently large number of study sites (Fig. 8.1) to allow both (a) accurate approximation of data using non-linear functions and (b) statistical comparison of the fit of non-linear and linear models to the data.

Although assumption on linearity may appear valid only for a relatively minor range of toxicant concentrations, it remains the metric of practical choice when the number of data points is relatively small and visual inspection of data suggests the absence of a dome-shaped or U-shaped pattern. Moreover, the linear correlation is one of the few metrics suitable for meta-analysis (Rosenberg et al. 2000), while there is no commonly accepted way to handle polynomial and other non-linear regressions.

However, when changes along pollution gradients are not monotonic, the use of linear correlation may appear misleading: both dome-shaped and U-shaped patterns may yield zero correlation even in the presence of strong association between pollution load and characteristics of biota. Importantly, some phenomenological models predict the existence of these patterns. In particular, the intermediate disturbance hypothesis proposed by Connell (1978) suggests that communities subjected to moderate level of disturbance may have greater species richness or diversity than both communities existing under benign conditions and those experiencing higher disturbance. In addition, the ‘humpback’ model predicts the strongest plant-plant facilitation at intermediate levels of stress (Rebele 2000; Maestre & Cortina 2004; Maestre et al. 2005; Gilad et al. 2007), at least when a resource stressor (drought or lack of nutrients) is the most important abiotic force affecting plant-plant interactions (Eränen 2009). On the other hand, survival selection proportional to pollution load may create a U-shaped pattern in some vitality indices: vitality is likely to be lowest at intermediate levels of pollution, where toxicity is high enough to adversely affect fitness, but too low to act as strong selecting agent enhancing population-wide tolerance by elimination of sensitive genotypes.

In our study, the second-degree polynomial regression fitted the data better than linear regression in 64 of 608 statistical tests, i.e., about twice as high as can be expected by chance at the accepted significance level $P = 0.05$. The ratio between dome-shaped and U-shaped patterns did not differ from 1:1 ($G_H = 0.066$, $df = 1$, $P = 0.79$). These results agree with the estimated frequency ($\leq 5\%$) of dome-shaped responses to pollution in published data sets on plant diversity (Zvereva et al. 2008) and density of insect herbivores (Zvereva & Kozlov 2009). The frequency of peaked

relationships between pollution load and diversity of vascular plants in original data (11.4%) is in agreement with the conclusion (Mackey & Currie 2001; Li et al. 2004) that diversity–disturbance relationships are peaked less frequently than predicted by the intermediate disturbance hypothesis.

We conclude that both dome-shaped and U-shaped dose–response patterns do occur in pollution gradients, but their frequencies are generally low. Therefore, correlation coefficients are likely to produce an unbiased estimates of the strength of relationships between pollution load and the investigated characters of terrestrial biota. However, proportions of dome-shaped and U-shaped patterns changed with investigated characters ($G_H = 31.4$, d.f. = 20, $P = 0.048$), hinting that the accuracy of estimates based on coefficients of linear correlation may vary across our study. More attention should be paid to identification of non-linear effects, in particular in association with exploration of causal relationships behind the observed phenomena (Section 9.2.3).

9.2.5 Gradualness of Responses

Terrestrial habitats that have been damaged the most by industrial pollution are associated with historical smelting sites, such as Trail, Wawa, Sudbery, and Karabash, suggesting that a long period of impact is necessary to cause substantial changes in an ecosystem. In a few well-documented situations, the expansion of industrial barrens lasted for decades (Kozlov & Zvereva 2007a). In combination with observations that ecological damage becomes progressively more severe as distance from the polluter decreases (Freedman 1989) and effects shown in large-scale experiments with both irradiation (Woodwell 1970) and industrial pollutants (Zobel et al. 1999) develop gradually, these data create an impression that the state of ecosystems experiencing pollution impact changes gradually over time.

We had no reason to question the validity of this gradualness assumption until we recognised that correlations between the duration of impact of the pollution and the magnitude of biotic effect occurred less frequently than expected. An analysis of published data only detected two significant relationships: adverse effects of pollution on species richness of woody plants increased with time (Zvereva et al. 2008), while adverse effects on insect performance decreased (Zvereva & Kozlov 2009). Furthermore, the oldest polluters demonstrated the strongest effects on soil microfungi in categorical analyses, but correlations between the magnitude of the effect and duration of impact appeared non-significant for both diversity and abundance characteristics (Ruotsalainen & Kozlov 2006). The failure to model temporal trends using linear functions demonstrates that changes in terrestrial biota around industrial polluters are often non-linear, although generally monotonic.

Catastrophic changes in ecosystems can be seen as extreme cases of non-linear responses to pollution. Changes of some factor may have little effect on an ecosystem until a threshold is reached, but then a shift to an alternative state occurs (Scheffer

& Carpenter 2003). These abrupt changes seem to be more common than earlier thought and evidence of catastrophic regime shifts in ecosystems has been increasingly reported in a variety of aquatic and terrestrial systems (Scheffer & Carpenter 2003; Genkai-Kato 2007).

It is difficult to experimentally prove that a system has multiple stable states (Scheffer & Carpenter 2003; Schroder et al. 2005), and previous observational evidence has mostly concerned aquatic ecosystems (Genkai-Kato 2007). Exploration of polluted ecosystems usually begins when a disturbance is already evident, therefore providing little or no information to judge the applicability of this theory to changes caused by pollution. We have only identified one study that illustrates changes in plant diversity in the first 3 years of a pollution event: near the Pyławy nitrogen plant (built in 1966) the number of vascular plant species decreased from presumably 23 (unpolluted control) to 13 in 1968 and two in 1969 (Sokołowski 1971). In contrast, repeated observations conducted around several polluters decades after their commencement revealed only minor changes in species richness with time (Zvereva et al. 2008), suggesting that major changes in diversity are abrupt rather than gradual.

The transition of polluted ecosystems to an alternative state is indirectly confirmed by observations conducted along pollution gradients. In the gradient starting at the copper smelter in Revda, the majority of the study plots were either in their original state or in a relatively stable impact state, suggesting that transition between these two states was very rapid (Vorobeichik 2003b). Similarly, we identified a clear spatial border between industrial barrens and forest ecosystems around Monchegorsk, with the transition zone rarely exceeding 100 m in width (V.E.Z. & M.V.K., personal observations, 2007). However, monitoring the recovery of industrial barrens near Sudbury did not provide support for the catastrophic regime shifts model (Anand et al. 2005).

We conclude that some data contradict an intuitive assumption on gradualness in ecosystem responses to pollution. On the other hand, few observations give support for the hypothesis that some ecosystems shifted to an alternative state following severe impact of industrial pollution and subsequently remained relatively stable in this new state. More long-term observations on both the decline and recovery processes are necessary to distinguish between these two alternatives - gradual changes and catastrophic shifts in polluted ecosystems.

9.3 Changes of Ecosystem Components Along Pollution Gradients: Structure of Phenomenological Model

Building a phenomenological model of ecosystem responses to industrial pollution requires linking observed effects with characteristics of both the polluters and impacted ecosystems. This task is complicated by multiple correlations between the investigated parameters, their generally weak relationships with pollution (Section 8.4.3), and the

importance of indirect and secondary effects (Section 9.2.3) that may mask or even counterbalance the direct effects of pollutants. To avoid redundancy, we first identified groups of parameters that demonstrated co-ordinated responses to pollution and selected a set of key parameters for further exploratory analysis. In the second stage, we explored the relationships between these key parameters and the characteristics of the polluters and the impacted ecosystems.

The exploration of the correlation structure of our data using principal component analysis, path analysis or structural equation models was prevented by an incomplete data matrix, resulting from the impossibility of measuring each parameter around all 18 polluters, structure of the data (effect sizes, not primary measurements), and absence of an *a priori* model. Therefore, we chose to identify correlation pleiades, or suites of highly correlated parameters that are relatively independent of other parameters within impacted ecosystems, using a maximum correlation path between our variables (Weldre 1964). The concept of correlation pleiades, suggested by Terentjev (1931), was introduced to the international scientific community by Berg (1960) and was then almost exclusively discussed in studies of plant morphology (Armbruster et al. 1999; Ordano et al. 2008).

The algorithm of the maximum correlation path analysis is straightforward (Weldre 1964; Schmidt 1984). The correlation matrix is searched for the largest (by absolute value) correlation coefficient, which denotes the first link between two characters. Then the correlation matrix is searched for the largest correlation between the two selected characters and all the remaining characters. The procedure is repeated until the last character is attached to the diagram. This algorithm identifies the single correlation path between any two variables, in contrast to diagrams produced by path analysis that allows multiple links between variables.

A stepwise regression analysis was used to search for the associations between changes in the key parameters of the investigated ecosystems, characteristics of the polluters, and the local climate. We used the following explanatory variables: mean January temperature, mean July temperature, annual precipitation (Table 2.2), duration of pollution impact (calculated as the difference between the year when the majority of data were collected [Table 2.25] and the year of the polluter's establishment [Section 2.2.2]), mean annual emissions of SO₂, metals, nitrogen oxides, and fluorine (primarily in the form of HF) during 2000–2004, and maximum documented annual emission of SO₂ since the beginning of pollution. The emission data were extracted from Tables 2.4–2.24.

The maximum correlation path diagram (Fig. 9.1) demonstrated that changes in the measured soil characteristics are more strongly linked with changes in biotic parameters than with each other. Furthermore, changes of soil pH and electrical conductivity, stoniness, and topsoil depth were each explained by different polluter characteristics (Table 9.1). These results indicate that several co-occurring processes modified soil characteristics around industrial polluters. It seems likely that pollution directly affected the soil pH, and changes in the soil pH resulted in a cascading effect on vegetation structure. Modification of vegetation, primarily the decline of top-canopy and understorey plants, caused a decrease in topsoil depth and an increase in stoniness, with a positive feedback to plant performance.

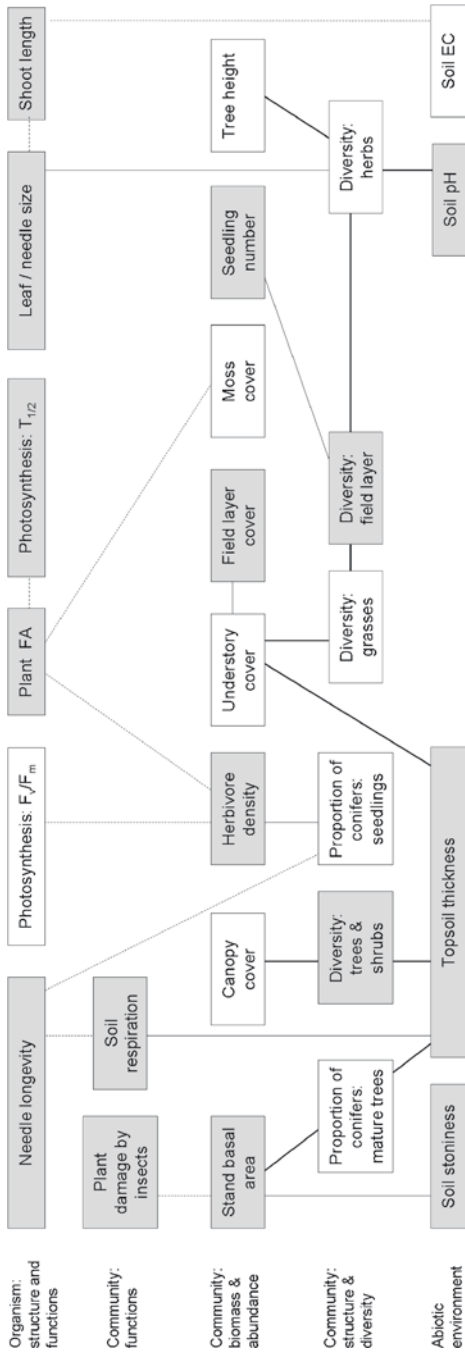


Fig. 9.1 The maximum correlation path diagram showing the strongest correlations between pollution-induced changes in the explored characteristics of terrestrial ecosystems. Thick lines, strong correlations ($|r| > 0.8$); thin lines, moderate correlations ($0.7 < |r| < 0.8$); dotted lines, weak correlations ($|r| < 0.7$). Shaded: characters, for which the sources of variation were explored using stepwise regression analysis (Table 9.1)

Table 9.1 Variables explaining variation in responses of the selected characteristics of terrestrial ecosystems to pollution

Group of characters	Response variable	Explanatory variable	Slope (mean ± S.E.)	Partial R ²	F (d.f.)	P
Abiotic environment	Topsoil depth	Duration of impact	-0.026 ± 0.007	0.70	14.1 (1, 7)	0.0095
	Soil stoniness	Mean emission of SO ₂ ^a	0.049 ± 0.016	0.39	9.33 (1, 6)	0.03
	Soil pH	Mean emission of metals ^a	-0.206 ± 0.034	0.77	36.0 (1, 11)	<0.0001
Community: structure and diversity	Species richness of field layer vegetation	Mean emission of metals ^a	-0.154 ± 0.024	0.53	11.5 (1, 11)	0.0068
		Mean July temperature	-0.178 ± 0.043	0.30	16.9 (1, 11)	0.0026
		Mean emission of metals ^a	-0.091 ± 0.042	0.67	4.75 (1, 9)	0.06
Community: biomass and abundance	Cover of field layer vegetation	Duration of impact	-0.016 ± 0.006	0.40	6.66 (1, 10)	0.03
	Density of birch-feeding herbivores	Mean July temperature	-0.084 ± 0.027	0.57	9.39 (1, 8)	0.02
	Soil respiration	Mean emission of metals ^a	-0.118 ± 0.028	0.60	17.8 (1, 12)	0.0012
Organism: structure and functions	Rate of photosynthetic reaction (T _{1/2})	Mean July temperature	0.074 ± 0.031	0.38	7.21 (1, 11)	0.02
	Leaf length	Mean emission of metals ^a	-0.065 ± 0.024	0.25	4.30 (1, 12)	0.06
		Mean July temperature	-0.082 ± 0.039	0.20	4.35 (1, 12)	0.06
	Shoot length	Mean emission of SO ₂ ^a	-0.082 ± 0.029	0.26	4.57 (1, 12)	0.05
	Needle longevity	Mean emission of SO ₂ ^a	-0.094 ± 0.047	0.47	8.00 (1, 10)	0.02
		Annual precipitation	0.006 ± 0.002	0.26	7.58 (1, 10)	0.02

Stepwise regression analyses included the following explanatory variables: mean January temperature, mean July temperature, annual precipitation, duration of pollution impact, mean annual emissions of SO₂, metals, nitrogen oxides, and fluorine (primarily HF) during 2000–2004, and maximum documented annual emission of SO₂ since the beginning of pollution. None of these variables entered the regression models at P = 0.06 for stand basal area, density of woody plant seedlings, plant damage by insect herbivores, and plant fluctuating asymmetry.

^aValues were log-transformed prior the analysis.

Changes in soil electrical conductivity were not explained by the amount of emissions and were only weakly associated with changes in terrestrial biota.

The lower respiration of polluted soils was primarily ($R^2 = 0.60$) related to higher metal emissions (Table 9.1). However, it seems likely that decreases in litterfall and soil moisture that occurred following the decline of top-canopy and understorey plants have also contributed to decreased soil respiration. Furthermore, a link between needle longevity and soil respiration (Fig. 9.1) denotes the strong influence of forest canopy processes on the activity of roots and associated organisms (Moyano et al. 2008).

We detected strong to moderate correlations between changes in measured characteristics of plant communities (Fig. 9.1). At the same time, the amount of emissions and duration of impact only explained a small part of changes in plant diversity and productivity (Table 9.1), suggesting that secondary effects modified the trajectories of the successions triggered by pollution (partially described in Section 9.2.3). Also, disturbances other than pollution, the history of which could not be accounted for in the course of our study (such as logging, fires, damage by pests and pathogens, and recreation), may have substantially contributed to modifications of vegetation structure.

Changes in parameters that reflect the state of long-lived woody plants (canopy cover, species richness of trees and large shrubs, stand basal area, and proportion of conifers among top-canopy trees) are only weakly linked to changes in understorey, field layer vegetation and moss cover (Fig. 9.1). Furthermore, no association was detected between changes of top-canopy vegetation and characteristics of pollutants. We suggest that the existence of a time lag between the onset of pollution impact and the decline of extant individuals in affected populations (Zverev et al. 2008; Zverev 2009) may mask the association between the characteristics of pollutants and changes in populations of long-lived plants.

Although magnitudes of organism-level effects were generally influenced by both the emission of sulphur dioxide and the local climate (Table 9.1), these effects only demonstrated weak links between one another and showed almost no correlation with community-level effects. Especially surprising is the absence of expected associations between changes in photosynthetic processes (measured by the efficiency of photosynthesis and the rate of photosynthetic reaction) and changes in plant growth processes (measured by leaf/needle size and shoot length). Similarly, changes in growth processes showed no strict correspondence with changes in plant abundance or biomass (Fig. 9.1). This pattern supports the conclusion (McClellan et al. 2008) that the results of organism-level studies cannot be directly translated into community-level effects.

Finally, we discovered weak negative correlations between changes in the abundance of birch-feeding insects, plant fluctuating asymmetry, and photosynthesis. At the same time, these indices all showed only weak associations with changes in other ecosystem characteristics. Variation in two of four indices from this group (herbivore density and $T_{1/2}$) is partially explained by mean July temperature, while the two other indices did not depend on either emission of pollutants or climate (Table 9.1). An exploration of the cause-and-effect relationships within this group of parameters

requires additional information, since existing data are insufficient to select one of two plausible models. Plant fluctuating asymmetry may increase (Zvereva et al. 1997; Møller & Shykoff 1999; Kozlov 2005c) and photochemical efficiency may decrease (Delaney 2008) following damage by herbivorous insects. On the other hand, low photochemical efficiency and increases in fluctuating asymmetry may indicate that plants experience abiotic stress (Daley 1995; Møller & Shykoff 1999; Nesterenko et al. 2007). Feeding on stressed plants may facilitate some guilds of herbivorous insects (Koricheva et al. 1998), leading to an increase in their population densities. More generally, the correlation structure of our data confirms Vehviläinen et al.'s (2007) conclusion that there are no consistent relationships between the abundance of insect herbivores and the diversity of forest trees.

Thus, we identified several groups of characteristics that showed coordinated responses to pollution. The similarity in responses of individual parameters may result from: (a) the structure of the data, (b) cause-and-effect relationships, and (c) a common cause behind several observed effects. To avoid redundancy, we restricted exploration of the sources of variation in responses to industrial pollution (Sections 9.4.1–9.4.3) to three of four characteristics of soils and to 13 of 22 community- and organism-level characteristics (Table 9.1), which, due to the correlation structure of our data (Fig. 9.1), are sufficiently representative of the entire suite of investigated parameters.

9.4 Sources of Variation in Biotic Responses to Pollution

9.4.1 *Variation Related to Emission Sources*

The links between the direction and magnitude of changes in different parameters of terrestrial ecosystems around individual polluters and characteristics of these polluters have remained almost unexplored. Even the comparisons between the classes of polluters have only yielded a few intuitive conclusions, like the smaller extent of vegetation damage observed around power plants relative to smelters and fluorine-emitting industries, and the less severe damage of terrestrial ecosystems around sources of fluorine-containing emissions relative to non-ferrous smelters (Bohne 1971; Freedman 1989). This is partially explained by the absence of comparative analyses in the vast majority of these primary studies.

Only four of 1,000 publications from our collection of primary studies (described in Section 8.4.1) compared the effects observed around four polluters, while 91% of publications reported observations conducted around a single polluter. Narrative reviews often considered the effects caused by different pollutants in different chapters (Scurfield 1960a, b, Alstad et al. 1982; Treshow & Andersen 1989; Heliövaara & Väisänen 1993; Flagler 1998). To our knowledge, a statistical comparison between the ecological effects caused by different classes of polluters was performed for the first time in a meta-analysis of soil microfungi data (Ruotsalainen & Kozlov 2006).

Effect sizes were first regressed to the emissions of sulphur dioxide by Zvereva et al. (2008). An analysis of published data demonstrated that the recorded biotic effects generally depend on the type of the polluter, and in some particular situations, also on the duration of the pollution impact (Ruotsalainen & Kozlov 2006; Zvereva et al. 2008; Zvereva & Kozlov 2009; Roitto et al. 2009).

In our study (Chapters 3–7), significant adverse effects of non-ferrous smelters were detected in 54% of individual meta-analyses (Table 9.2), i.e., much more frequently than around aluminium smelters (11%). These results support the conclusion by Freedman (1989) that aluminium industries generally impose weaker effects on terrestrial biota than non-ferrous smelters. However, the generally weak effect of aluminium smelters may be enhanced in some specific situations, especially in warmer and more humid climates (discussed in Section 6.4.3).

The proportion of significant negative effects detected by meta-analyses was highest (62%) among polluters that caused soil acidification. This result, although in agreement with earlier conclusions that acidification causes strong detrimental effects (Abrahamsen 1984; Roem et al. 2002; AMAP 2006), cannot be attributed to acidification alone, since most acidic polluters also emit heavy metals. Similarly, the relatively high (22%) occurrence of negative effects in the absence of changes in soil pH can most probably be explained by the emission of heavy metals by three of five polluters from this category (Harjavalta, Norilsk and Sudbury). Minor effects of Norilsk on soil pH are primarily due to the high buffering capacity of soils (AMAP 2006), while the absence of significant changes in soil pH around Harjavalta and Sudbury resulted from the substantial decline of sulphur dioxide emissions over the past decades. Alkaline polluters affected about the same proportion of variables as neutral polluters, but caused both negative (17%) and positive (8%) effects (Table 9.2).

A stepwise regression analyses conducted to determine the variables associated with the variation in responses of 16 selected ecosystem parameters across the polluters yielded 12 significant models (Table 9.1). Amount of emissions was chosen by the program as explanatory variable in eight of these 12 models. This result seems predictable, since polluters producing more emissions are likely to cause stronger environmental effects. However, in similarly performed analyses of published data, the amount of emissions was not selected in any of the eight significant regression models (Zvereva et al. 2008; Zvereva & Kozlov 2009), although these regression analyses had larger statistical power due to larger sample sizes. We suggest that the absence of association between biotic effects and amount of emissions in earlier analyses may have resulted from research and publication biases that have affected the outcomes of primary studies. More specifically, elimination of small and non-significant effects at both the recording and publication stages (discussed in Section 8.5) may have created a false impression that the effects of pollution are always strong and significant (discussed in Section 9.2.2), and thus the magnitude of effects appeared to be independent of the strength of pollution impacts. On the other hand, emission data for the polluters explored in our study are more complete and correspond better to the timing of data collection than for the polluters, the effects of which were summarised in earlier meta-analyses. The discovery of dose-effect relationships

Table 9.2 Summary of the effects revealed by meta-analyses

Group of characters	Character	Sources of variation										Geographical position	
		Polluter types		Impact on soil pH									
		Overall effect	Non-ferrous smelters	Aluminum smelters	Acidic	Neutral	Alkaline	North	South				
Abiotic environment	Topsoil depth	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Soil stoniness	+	+	+	+	+	+	+	+	+	+	+	+
Community: diversity	Soil pH	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Soil electrical conductivity	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
	Species richness of trees and large shrubs	NS	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
	Species richness of herbs	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Species richness of grasses	NS	NS	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Species richness of field layer vegetation	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Species richness of all vascular plants	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Proportion of conifers in a stand	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Proportion of conifers among seedlings	NS	NS	NS	NS	NS	NS	(no data)	NS	NS	NS	NS	NS
	Cover of top-canopy trees	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
Community: biomass and abundance	Cover of understory	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Cover of field layer vegetation	-	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Cover of mosses	-	NS	-	-	NS	NS	NS	NS	NS	NS	NS	NS
	Stand basal area	-	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Stand height	-	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Number of seedlings of woody plants	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
	Density of birch-feeding herbivores	NS	NS	NS	NS	NS	NS	(no data)	NS	NS	NS	NS	NS
	Soil respiration	-	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Plant damage by herbivorous insects	-	NS	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Efficiency of photosynthesis (F_v/F_m)	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Organism: structure and functions	Rate of photosynthetic reaction ($T_{1/2}$)	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
	Leaf/needle size of woody plants	NS	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
	Shoot length of woody plants	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
	Needle longevity	-	-	NS	-	NS	NS	NS	NS	NS	NS	NS	NS
Fluctuating asymmetry of woody plants	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	

Only analyses based on correlation coefficients are included. Plus (+), values increased with pollution; minus (-), values decreased with pollution; NS, values demonstrated no significant changes with pollution.

further stresses the need to provide sufficient information about polluters when describing their impacts on organisms and ecosystems (Section 8.4.6).

Medical studies often link health effects with past air pollution levels (Moffatt et al. 2000; Rosenlund et al. 2006; Kohlhammer et al. 2007). However, the maximum documented annual emissions of SO₂ since the beginning of pollution did not enter any regression model (Table 9.1), while recent emissions of sulphur dioxide and metals entered three and five of 12 significant regression models, respectively (Table 9.1). The average slope of regression models that linked effect size with log-transformed values of annual emissions of heavy metals was twice as high as for emissions of sulphur dioxide (mean ± S.E.: -0.142 ± 0.025 vs. -0.075 ± 0.013 , respectively). These results indicate a higher contribution of metal emissions to changes observed around point polluters relative to sulphur dioxide.

The duration of impact entered only two of 12 significant regression models (Table 9.1). The increases in adverse effects on topsoil depth and cover of filed layer vegetation with time are consistent with the effects of the duration of the impact on diversity of vascular plants detected by meta-analysis of published data (Zvereva et al. 2008). The rarity of associations between duration of impact and magnitude of effect among both published and original data may hint at either the non-linearity of responses (Section 9.2.5) or the adaptation to pollution through either physiological acclimation or selection for pollution resistance. The adaptation hypothesis was only confirmed so far for field-collected arthropods, the performance of which improved with the time elapsed since the beginning of the impact (Zvereva & Kozlov 2009).

The relatively low number of data points used in the regression analyses (Table 9.1), as well as the low number of parameters affected by the amount of emissions, resulted in the low accuracy of our threshold emission level estimates. Uncertainties are also associated with the impossibility of separating the effects of sulphur dioxide and metals using our data set, where emissions of these two groups of pollutants strongly correlate to each other ($r = 0.63$, $N = 18$ polluters, $P = 0.0049$). The model calculation suggests that annual emissions that do not exceed 10 t of metals or 350 t of sulphur dioxide are unlikely to cause any changes in terrestrial ecosystems surrounding point polluters. On the other hand, emissions exceeding 390 t of metals or 595,000 t of sulphur dioxide will obviously result in detectable biotic effects. This result agrees with our earlier estimate that polluters emitting <1,500 t of SO₂ annually are unlikely to cause depauperation of plant communities (Zvereva et al. 2008) and further supports the conclusion (Freedman 1989; Kozlov & Zvereva 2007a; Zvereva et al. 2008) that the effects of industries emitting sulphur dioxide but not metals are generally less detrimental than effects of non-ferrous smelters.

The amounts of fluorine-containing emissions were reported for eight of our polluters (six aluminium smelters, the copper smelter at Revda, and the cement plant at Vorkuta), while the amounts of sulphur dioxide released into the ambient air were known for all 18 polluters. Since fluorine is at least as toxic as sulphur dioxide (Bohne 1971; Treshow & Andersen 1989; Flagler 1998; Meldrum 1999), the question arises whether the low number of fluorine-emitting industries in our sample is the primary reason for not detecting associations between fluorine emissions and

changes in ecosystem characteristics. To partially test this hypothesis, we calculated the correlations between all investigated parameters and emissions of fluorine across six aluminium plants. A single correlation was found to be significant at the table-wide probability level $P = 0.05$, namely: the needle longevity decreased with increases in fluorine emissions ($r = -0.99$, $N = 4$ polluters). The coefficients of determination (R^2) for sulphur dioxide and fluorine, averaged for these six aluminium smelters across 27 measured parameters, were nearly identical (0.22 and 0.23, respectively). This result is consistent with strong positive correlations between the amount of fluorine and sulphur dioxide released by aluminium smelters ($r = 0.78$, $N = 6$ polluters, $P = 0.07$). Thus, although we do not think that sulphur dioxide is the primary cause of the biotic effects observed around aluminium smelters, still these effects can be predicted on the basis of sulphur dioxide emissions.

Nitrogen oxide emissions did not enter any regression model. This result is in line with the commonly accepted opinion about the low contribution of nitrogen oxides to local effects observed around point polluters (Heliövaara & Väisänen 1993; Flagler 1998; Bytnerowicz et al. 1999). Nitrogen oxides may even act as fertilisers in some situations and actually improve plant growth in soils deficient in nitrogen (Abrahamsen 1984; Treshow 1984). However, excessive emissions of nitrogen-containing dust may cause acute damage, including forest decline around fertilizer producing factories (Sokołowski 1971; Kowalkowski 1990). Nitrogen oxides are indeed important at regional scales, due to both nitrogen saturation effect and a contribution to ozone formation (Lee 1998; Bytnerowicz et al. 1999; Karnosky et al. 2003a; Hyvönen et al. 2007; Hettelingh et al. 2008).

To conclude, changes in ecosystem parameters around industrial polluters generally depend on both composition and amount of emissions, but are only rarely associated with the time since beginning of pollution. Acidic polluters cause stronger effects on biota than alkaline polluters; the effects of metal-emitting industries are generally more detrimental than the effects of polluters only emitting sulphur dioxide.

9.4.2 Variation Related to Impacted Ecosystems

It became evident long ago that the same pollution loads may cause different effects in different ecosystems and/or regions. Differential sensitivities to pollutants were formalised through the concept of critical loads (Nilsson & Grennfelt 1988). The critical load for a pollutant defines a deposition level below which sensitive parts of ecosystems are not affected. A critical load approach has been widely and successfully used to evaluate the effects of acid precipitation on ecosystems in Europe (Nilsson & Grennfelt 1988; Posch et al. 2001) and have shaped air-pollution control policies since the 1980s (Burns et al. 2008). Reinds et al. (2006) published a preliminary assessment of critical loads for several metal pollutants, including copper and nickel. However, we are not aware of any attempt to link the strength of biotic effects caused by point polluters with the existing estimates of critical loads for the impacted areas.

To ensure the uniformity of data on critical loads, we restricted the following analysis to territories covered by the European assessment programs (Posch et al. 2001; Reinds et al. 2006). We extracted data on the maximum critical load of sulphur, nickel and copper from the maps published in previously mentioned reports, and extrapolated these maps to obtain approximate measures of critical loads for two polluters situated in Ural, i.e. slightly outside the mapped region.

The addition of critical loads of sulphur to the list of explanatory variables did not change the results of the stepwise regression analyses described in Section 9.4.1, while critical loads for nickel entered three regression models. The adverse effects of pollution on soil pH, species richness of field layer vegetation, and leaf/needle size were less expressed in regions with higher critical loads. This result seems to agree with the conclusion that metals contribute to adverse effects more than sulphur dioxide. However, metal emissions were only reported for eight of the 18 polluters included in our analysis, and nickel was the major metal pollutant for only two of these eight polluters. At the same time, the critical loads for copper, which was emitted by a larger number of polluters and in generally larger quantities, entered only one regression model, but with the opposite sign than expected. The adverse effects of pollution on shoot length were stronger ($R^2 = 0.30$) in regions with higher critical loads of copper. This gives us a reason to suspect that the detected correlations with critical loads for metals are spurious. This hypothesis was confirmed by stepwise regression analyses restricted to the eight metal-emitting polluters, which demonstrated a much lower (instead of the expected higher) explanatory power of the critical load for metal pollutants relative to the analyses based on all 18 polluters (data not shown). Our results may hint that uncertainties in input parameters used to calculate the critical loads for Europe are larger than commonly thought.

Another aspect of ecosystem sensitivity to disturbances is linked to ecosystem structure and functional organisation. In particular, relationships between species diversity and ecosystem stability, that have been and still are widely debated, may have direct implications for understanding how communities respond to pollution and other anthropogenic stressors (Clements & Newman 2002). While four models reviewed by Peterson et al. (1998) assume a positive relationship between stability and diversity, an opposite pattern was both predicted theoretically (May 1973; Lehman & Tilman 2000) and detected in several experiments (Clements & Newman 2002; Foster et al. 2002).

A meta-analysis of published data (Zvereva et al. 2008) demonstrated that pollution effects on the species richness of vascular plants became more severe with increases in the diversity of the impacted communities. To verify this hypothesis with original data, we conducted a stepwise regression analysis using the mean numbers of the species with the two most distant (unpolluted) sites added to the explanatory variables. A strong negative association ($R^2 = 0.68$) was only detected between the magnitude of the pollution effect and regional diversity for herbs. Among other characteristics of the plant community, only stand basal area showed a moderate negative association ($R^2 = 0.45$) between effect size and regional diversity of vascular plants. These results are in agreement with the overall increase of

adverse effects with summer temperatures (Section 9.4.3), which can be seen as a sign of the higher stability of Northern, generally less diverse, ecosystems.

To conclude, little is known about the ecosystem properties that determine their sensitivity to the impacts of industrial pollutants. Critical loads of sulphur, nickel, or copper did not explain the variation in responses of the investigated components of terrestrial ecosystems to pollution, while a lower diversity of plant communities was associated with smaller changes in some characteristics of vegetation around industrial polluters.

9.4.3 *Variation Related to Climate*

Although environmental pollution is an integral part of global change (Taylor et al. 1994), the vast majority of the research addressing the biotic effects of global change has overlooked the pollution issue (but see Settele et al. 2005). On the other hand, extensive studies on both the distribution of pollutants and the biotic effects of pollution have generally neglected climate effects, even though changes in the toxicity of pollutants as a result of temperature changes were already known several decades ago (Cairns et al. 1975). The interactive effects of air pollution and temperature were documented in a number of studies on plant growth (Alekseev 1991; Shevtsova 1998), different characteristics of aquatic biota (Sokolova & Lanning 2008), and human health (Mugge 2007; Hu et al. 2008, and references therein).

Recent scientific assessments (Houghton et al. 2001) and innovative integrated projects (European Environment Agency 2003) started to change the situation, bridging research fields that for a long time have developed independently. Simultaneously addressing air pollution and climate change problems would potentially result in better integration of local, national and global environmental policies (Swart et al. 2004; Bytnerowicz et al. 2007).

Our results confirm latitudinal variations in the effects of industrial pollution on biota that were previously revealed in published data on plant diversity (Zvereva et al. 2008). In 8 of 23 individual meta-analyses (Table 9.2), adverse effects were significant only around either northern or southern polluters. This variation can be attributed to variations in both diversity (discussed above, Section 9.4.2) and climate. The discovery of the climatic variation of responses of terrestrial arthropods to industrial pollution, which was consistent with the climatic variations observed in plant communities, allowed us to suggest that the enhancement of pollution effects in warmer and moister climates concerns many aspects of ecosystem structure and functioning (Zvereva & Kozlov 2009). This hypothesis is confirmed by our study of point polluters. Climatic data entered five of 12 significant regression models, consistently demonstrating that adverse effects of pollution on community- and organism-level parameters increased with increases in mean July temperatures (Table 9.1).

The lower sensitivity of high-latitude ecosystems to the impact of industrial pollution, revealed by the analyses of both original and published data, can be explained by several reasons. First, lower temperatures may result in lower stomata

opening, decreasing the impact of gaseous pollutants on plants (Bytnerowicz et al. 1999). Second, the shorter vegetation period naturally decreases the exposure to pollutants. Third, high-latitude soils generally contain fewer nutrients, and therefore the adverse effects of toxic pollutants may be counterbalanced to some extent by the fertilising effect of nitrogen oxides (Abrahamsen 1984; Treshow 1984). Finally, higher temperatures and increased precipitation may enhance mobility and increase toxicity of pollutants (Cairns et al. 1975; Klein 1989; Tipping et al. 1999), explaining the generally higher adverse effects seen in a warmer climate.

Only a few attempts have been made to predict the consequences of climate change for polluted areas. At the organism level, the effects of temperature on the sensitivity of Norway spruce and European beech to sulphur dioxide and nitrogen oxides were explored using a model of stomatal conductance. Although the expected responses to temperature increases depend on both species and region, a simulation showed that the sensitivity of trees to air pollution would increase in boreal areas of Europe (Guardans 2002). At the community level, pollution effects on plant diversity are likely to increase in warmer climates (Zvereva et al. 2008). Also the decline of arthropod decomposers and predators and increase in herbivory with increases in pollution become more severe with increases in mean temperature and precipitation (Zvereva & Kozlov 2009). All these effects are likely to have adverse consequences for ecosystem-level processes.

Thus, the effects caused by industrial pollutants critically depend on climate. Although our results are in contrast to the predictions based on the critical loads concept (Posch 2002), they provide support to the hypothesis (Holmstrup et al. 1998) that under a warmer climate, existing pollution loads may become more harmful. This is especially important for Northern Europe and some regions in the USA and Canada, which have been (and in some places still are) affected by emissions from major non-ferrous smelters.

9.5 Exploring Effects of Industrial Pollution: Prospects and Limitations

The need to integrate the knowledge of biotic effects caused by pollution is obvious. Several recent reviews (Percy et al. 1999; Clements & Newman 2002; Adams 2003; Cape et al. 2003; Ferretti et al. 2003; Karnosky et al. 2003c; Percy & Ferretti 2004; Paoletti et al. 2007) have stressed the need to develop pollution ecology through the concerted effort of scientists who monitor the effects of pollution, explore them by means of field and laboratory experiments, and model both distribution of pollutants and ecosystem processes. These reviews also identified critical data gaps and research needs in several specific fields of knowledge.

Over the past decade, the focus of pollution ecology largely shifted from the acute localized effects of industrial pollutants, such as sulphur dioxide and heavy metals, to the regional effects of ozone and nitrogen deposition. In many industrialised

countries, ‘traditional’ pollutants have dropped out of priority lists, particularly in forestry-oriented studies (Ferretti et al. 2003; Paoletti et al. 2007). Under these circumstances, our research may look outdated due to its focus on point polluters. However, the annual symposium of the British Ecological Society in 2008, entitled ‘Ecology of Industrial Pollution’, attracted delegates from universities, industry, government agencies, museums and consultancy firms, demonstrating the existence of sufficient interest to this problem (Crowden 2008). On the other hand, research funds for studying the effects of ‘traditional’ pollutants have dwindled considerably in the last decades, leading to the pessimistic conclusion that “progress on these issues will come slowly if at all” (Lovett & Tear 2007).

We are a bit more optimistic. We think that our study not only identified the acute need for a better understanding of the link between sources of industrial pollution and effects on structure and functions of terrestrial ecosystems, but also revealed the possibility of achieving this goal within a reasonable time frame.

First, and most importantly, we demonstrated that it is possible to build a phenomenological model linking changes in key parameters of terrestrial ecosystems with pollution data. However, when building this model, we need to appreciate that the effects caused by pollution are generally low to moderate, and even in the impact zones of large industrial enterprises they can be masked by the effects of both natural variation and other disturbances.

Second, we revealed substantial variation in biotic responses to pollution related to both characteristics of polluters, characteristics of impacted ecosystems, and climate. Our inability to reveal a consistent and significant effect of ‘pollution in general’ on most of the explored characteristics results not from the absence of responses, but from their diversity, often resulting in a zero overall effect. Thus, rough parameterisation of the most important links within this model is only possible when identified sources of variation are accounted for.

Third, since the links between the organism-level and community- or ecosystem-level changes appeared weak, the impact of pollution on ecosystem-level processes should be explored directly rather than deduced from organism-level studies.

The shortage of funding for field studies of biotic effects caused by industrial pollutants emphasizes the importance of careful selection of the topics of the planned research. In our opinion, the top-priority list includes the following research areas, the exploration of which is of critical importance for large-scale generalizations in the field of pollution ecology:

Study objects:

- Geography: Africa, South-Eastern Asia, South America, Oceania
- Biomes and ecosystems: tropical forests, deserts, grasslands, savannah, tundra
- Industries: power plants, chemical industries, small polluters

Research topics

- Belowground processes
- Changes in species interactions
- Functional diversity of communities

- Spatial structure and demography of populations
- Ecosystem-level effects

We also suggest that pollution ecologists expand the use of the following research approaches:

- Comparisons between polluters, organisms, communities, and ecosystems
- Long-term studies
- Exploration of dose-and-effect and cause-and-effect relationships
- Research synthesis

Finally, pollution ecology cannot develop in isolation. Bridging the gaps between pollution ecology, basic ecology, ecotoxicology, and global change research is critically important for advancing the development of all these disciplines. The experience accumulated during the integrated research of ozone impact on terrestrial biota (Karnosky et al. 2003a, b; Hayes et al. 2007; Marinari et al. 2007) clearly shows the existence of scientific potential for rapid collection of high-quality information, which is necessary to build the predictive models that account for multiple interactions between pollutants, ecosystem properties, and climate.

9.6 Summary

The adverse consequences of pollution impacts on terrestrial ecosystems have been under careful investigation for more than a century. However, we are still far from making large-scale generalizations. This is primarily due to the insufficient quality of many studies, prejudgement, and misinterpretation of the results. Changes occurring in organisms, communities, and ecosystems affected by industrial pollutants are neither general nor uniform. Identification of factors explaining the diversity in responses is crucial for understanding the links between sources of industrial pollution and effects on structure and functions of terrestrial ecosystems. Changes in investigated parameters of biota generally depend on both composition and amount of emissions, but are only rarely correlated with the time since the beginning of pollution. Acidic pollutants cause stronger effects on biota than alkaline pollutants; effects of metal-emitting industries are more detrimental than the effects of pollutants emitting only sulphur dioxide. The detrimental effects are stronger in Southern ecosystems and depend on summer temperatures in such a way that in a warmer climate the existing pollution loads may become more harmful. Information accumulated by pollution scientists allows for the construction of a phenomenological model that links changes in key parameters of terrestrial ecosystems with pollution and climate data, and for a rough parameterisation of the most important links within this model. Uncertainties in this model can be reduced by summarising outcomes of earlier research by means of meta-analysis, searching for factors explaining variation across the primary studies, and designing new projects to cover areas neglected previously by pollution scientists.