Modeling of vegetation diversity and a national conservation planning: example of Russia

Dissertation

for obtaining academic degree
Doktor der Naturwissenschaften
(Dr.rer.nat.)

in Ecology

submitted to
Faculty of Mathematics and Sciences
of the University of Potsdam

by
Irina Venevskaia

Potsdam, October 2004

CONTENTS

PAPERS:

Introduction

Management of biodiversity on a national level often does not coincide with international efforts and even on an international level a limited consensus has so far been achieved on biodiversity conservation priorities (Mace et al., 2000). The duplication of investigation and management efforts on the international and national level, leading to competition instead of complementary priority setting, can be explained to certain extent by conceptual difficulties in biodiversity conservation (see discussion in Pimm and Lawton, 1998). The necessity to produce a clear and practical strategy for biodiversity conservation, which is necessary to guide decision-makers on international and national levels, is widely recognized (Mace et al., 2000); (Fonseca et al., 2000) and is now being discussed by scientific community.

Considerable progress has been achieved recently in developing systematic conservation planning principles (see review in Margules and Pressey, 2000) for realisation of such a strategy. The geographical objects of systematic conservation planning should be existing or prospective sets of protected areas, which are supposed to represent or sample biodiversity, ideally on all levels of biological organisation, and to promote long-term persistence of elements of biodiversity (ecosystems, species or populations).

Overcoming shortcomings in the existing reserve networks in a region needs a systematic iterative approach (Margules and Pressey, 2000), consisting of four analysis and two management stages, respectively. The analysis stages include measuring and mapping biodiversity; setting quantitative conservation targets; and reviewing existing protected areas based on the targets and selection of additional conservation areas.

These analyses stages of systematic conservation planning were recently successfully applied for the large-scale conservation plan in South Africa (see review of the CAPE project in Cowling et al., 2003).

Systematic national conservation planning should operate with surrogate measures of biodiversity similar to the global ones in order to harmonise intra-national and international conservation strategies, recommended by the Convention on Biological Diversity (UNEP, 1992). Such a surrogate measure, combining representative concentrations of species and a level of habitat destruction, is applied within the “global biodiversity hotspot” approach (Myers et al., 2000).

Using this approach, 25 regions with exceptional species diversity and under considerable human pressure were defined as “global biodiversity hotspots” for global conservation by Conservation International (Myers et al., 2000). The hotspot boundaries were mapped using the principle of `biological commonalities`, i.e. each area features a separate biogeographic unit either apparent from its geographical location (like the islands of New Zealand, Madagascar etc) or reflecting the best-judgment of field experts (e.g. Tropical Andes and Mesoamerica) on regional flora. To qualify as a global hotspot, an area must contain at least 0.5% of the world’s 300 000 plant species as endemics, and should have lost 70% or more of its primary vegetation (Myers et al., 2000).

Ideally the ‘hotspots’ approach for mapping biodiversity, continued with further stages of systematic conservation panning, could provide a consice strategy for national conservation planning.

However, studies relating national conservation planning and international conservation planning are practically absent. Several unresolved issues are hindering realization of such studies:

- Basic biotic data for mapping vegetation diversity does not exist in many countries or is too scarce, while abiotic data is available

- A methodology relating to criteria measures for global and national biodiversity hotspots is absent
- Concepts and algorithms for setting conservation targets which underpin the difference in environmental conditions and human threats between global or national biodiversity hotspots, are not elaborated
- Only a few national conservation plans reflect the hierarchical nature of biodiversity (CAPE plan, Cowling et al., 2003).

Objectives

The overall objective of the study is *to elaborate quantitative methods for national conservation planning, coinciding with the international approach ('hotspots' approach).*

This objective requires solution of the following problems:

1) How to estimate large-scale vegetation diversity from abiotic factors only?
2) How to adopt the "global hotspots" approach to define the borders of national biodiversity hotspots?
3) How to set conservation targets which take account of the difference in environmental conditions and human threats between national biodiversity hotspots?
4) How to design a large-scale national conservation plan reflecting the hierarchical nature of biodiversity?

The case study for national conservation planning is Russia. Despite Russia having the largest area of any country in the world, national biodiversity studies are still poorly represented in the international scientific literature.

Summary of the papers

Paper 1 Venevsky, S.,Venevskaia, I., 2003. Energetic and landscape factors of global vegetation diversity. Ecology Letters 6, 1004-1016

It is shown that at the large scale, the species number of vascular plants can be predicted to a large extent by the climatically determined latent heat for evaporation and the geometrical structure of landscape, described as an altitudinal difference. Application of the energy-equivalence rule across plant communities for transpiration per area unit, and use of the fractal theory for the description of habitat occupation by vegetation results in a physically-based species-energy relationship.

This relationship allows the number of species of vascular plants to be estimated for different scales from the climate variables (monthly temperature and monthly precipitation), as long as values of the fractal dimension of vegetation fragmentation and lacunarity of landscape are known.

Application of averaged global constants for the fractal dimension of vegetation fragmentation and lacunarity of landscape in the relationship and correction for geometrical structure of landscape, described as an altitudinal difference, results in a species-area equation with known parameters.

Despite its simple form, this species-energy relationship generally reproduces global patterns of vegetation diversity, described as species number of vascular plants (SNVP), for scales

10 000 and 100 000 km^2 and is applicable for different regions across scales from one hundred to one million km^2.

The correlation between the theoretical and observed species numbers of vascular plants for global data sets are: r^2 =0.77 for the sampling area of 10 00 km^2 and r^2=0.76 for 100000 km^2.

The computed values for SNVP were also correlated with the observed data from botanical surveys in different vegetation zones, from arctic tundra to the subtropics. The data includes 59 observations from various authors for different area sizes, ranging from 100 km^2 to 4000 km^2, in Europe, Asia, Australia, Africa and North America. Despite the variety of vegetation zones, geographical locations and area sizes, the observed and calculated number of vascular plant species is strongly correlated (r^2=0.81, F=241, P<$0.2*10^{-22}$, slope=1.03).

To investigate the upper scale limitations of the species-energy relationship based on global constants, we analysed the survey data for countries and geographical provinces with significantly different species abundance, ranging from 0.8 to 16 million km^2. The correlation between observed and theoretical species number of vascular plants per 1 m^2 ($r^2$2=0.83, F=70, P<0.0000076, slope=0.66) averaged over the 16 regions proves that the model can be applied for scales up to several million kilometres.

The proposed theory produces more robust results for Wallace's phytogeographical provinces compared with the correlated-based approaches (Francis and Currie, 2003). The final climate-richness equation for vascular plants has a simple analytical form in comparison with formulations, suggested by the correlated-based approaches (see O'Brien, 1993; O'Brien, 1998; Francis and Currie, 2003) and requires easily measurable variables.

Paper 2 Venevsky, S.,Venevskaia, I., 2004. Hierarchical systematic conservation planning for a national level: example of Russia. 1. Identifying national biodiversity hotspots using abiotic factors. Biological Conservation (accepted)

National conservation planning should operate with measures of biodiversity similar to the global ones in order to harmonise intra-national and international conservation strategies. We suggest quantitative measures which allow transfer of two criteria of the 25 global biodiversity hotspots (Myers et al., 2000) to a national level for 74 large countries, and show how these measures can be applied for mapping national biodiversity hotspots. To qualify as a global hotspot, an area must contain at least 0.5% of the world's 300 000 plant species as endemics, and should have lost 70% or more of its primary vegetation (Myers et al., 2000).

The plant endemism criteria of global hotspots are captured by quantitative measures of endemism, which are approximately scale-independent and can be corrected for a country's environmental conditions and priorities in conservation. The definition of a global biodiversity hotspot is based on a type of the Threshold Endemism (TE) measure: TE is equal to the percentage of plant endemic species of the world's 300,000 plant species, where endemics can be met only in this hotspot and nowhere else (Myers et al., 2000). Such a definition of TE does not take into account the area of a hotspot and, thus, can be applied at national scale.

The plant endemism criteria for a national biodiversity hotspot can be defined using the TE lower limit approach as for the global hotspots: a region in a country is defined as a national biodiversity hotspot if the TE of the region is larger than a certain predefined percentage of the total number of the country's plant species.

The lower limit of TE, applied for definition of a national hotspot, should explicitly account for the differences in the total number of plant species found in the country and on the globe.

Indeed, the country's total number of plant species is influenced by apparent climatic, edaphic and geological conditions. Hence, the lower limit of TE should be increased for countries with poor plant species richness and decreased in the case of rich flora in order to balance the relative cumulative size of national hotspots with their relative global biodiversity value. We suggest correcting the lower limit of TE in defining a national hotspot proprtionally to the ratio of the average global number of vascular plant species per area unit to the average number of vascular plant species per area unit in a country. The last two variables can be obtained either from botanical surveys or from the species-energy relationship for vascular plants (see Paper 1).

The Threshold Endemism measure for a national biodiversity hotspot is logically connected with the Weighted Endemism (WE) (Williams, 2000) and Corrected Weighted Endemism (CWE) (Crisp et al., 2001) measures, which conceptually retain continuity in the quantitative endemism definition. We showed that the CWE for the global and national hotspots in our definition is proportional to the ratio of endemic plant species in a hotspot. By comparison of the minimum, maximum and average value of the CWE measure for 12 Australian biodiversity hotspots (Laffan and Crisp, 2003) with the similar values for the ratio of endemics in the 25 global hotspots (Myers et al., 2000) we found that the CWE will be approximately equal to the ratio of endemics in national hotspots.

Analysis of the CWE for the Australian hotspots and the ratio of endemics in the global hotspots reveals that:

- moving from the global scale to the scale of a large country (large than 200,000 km^2) will not significantly change the average CWE or the average ratio of plant endemics
- the ratio of endemics in large-scale national hotspots will vary similarly in both range and average for countries larger than 200,000 km^2, as it does for the global hotspots;
- at large scales one should expect a slowing in the increase of the ratio of endemic plant species in a national hotspot, when the area of this hotspot is gradually increased.

Therefore, as the first approximation we can adopt the global average values for the ratio of plant endemics, 0.43 (the average for global continental hotspots (Myers et al., 2000)) and 0.52 (the average for the global hotspots) as surrogates for the average ratio of plant endemics in national hotspots for 74 of countries which are larger than 200,000 km^2.

This approximation allows us to substitute estimates of the number of endemic plant species in a national hotspot by the total number of vascular plant species, which can be calculated from abiotic factors, using our theory (see Paper 1).

The flexible land use criteria for national biodiversity hotspots are defined from the percentage of natural vegetation remaining in the global hotspots.

Together agricultural lands and urban areas cover nearly 35% of the terrestrial land (Foley et al., 2003). The percentage of land use conversion of natural vegetation in the hotspots is at least two times the global value. We can use a similar criteria for identification of national hotspots in a country: an area should have lost at least two times more primary vegetation in comparison with the country's average loss as a whole.

Thus, we show that national biodiversity hotspots can be mapped from the species-energy relationship for vascular plants using climate, topographical and land use data, when the spatial pattern of species richness is unknown.

The elaborated methodology for mapping national biodiversity hotspots from abiotic factors was applied for the case study Russia. Four initial widely-separated grid cells of 0.5°x0.5° with the strongest endemism attributes, i.e. the highest SNVP zone, near large water bodies, and with the highest altitude difference, were chosen. The Russian climate is strongly influenced by the long-term quasi-stationary Siberian High with a characteristic linear scale 2000-3000 km, so we set initial points with a distance of approximately 2000 km from each other.

The minimum-area-required approach (to locate the minimum area required for the threshold number of endemic plant species, calculated using SNVP) was applied to define the borders of national biodiversity hotspots from the simulated vascular plants species richness data at 0.5°x0.5° spatial resolution.

Three Russian biodiversity hotspots, North Caucasus, South Siberia and the Far East were identified, comprising approximately 3% of the entire country area. The resulting hotspots cover national-scale environmental gradients in Russia (Stolbovoi and McCallum, 2002) and are also identified by Russian experts, but without specifying actual areas (Ministry of National Resources, 2002).

Paper 3 Venevskaia, I.,Venevsky , S., 2004. Hierarchical systematic conservation planning for a national level: example of Russia. 2. Setting conservation targets for Russian biodiversity hotspots. Biological Conservation (submitted)

The aim of this paper is to set conservation targets for species found in the three Russian biodiversity hotspots, North Caucasus, South Siberia and the Far East in terms of the number of locations where species occur.

Species in the Russian Red Data book in the five taxa used in global biodiversity hotspot analysis (terrestrial mammals, amphibian, reptiles, birds and vascular plants) are selected for setting conservation targets in the hotspots. The rationale here is that data on rare, threatened and endangered species, listed in Red Data books (RDB), have commonly been retained for setting and implementation of conservation priorities on a national level (for USA, Abbitt et al., 2000, for South Africa, Cowling et al., 1999). Indeed, this biodiversity feature by definition is already at significant risk of loss and, therefore, requires immediate protection measures. Besides, components of the RDB classification explicitly or implicitly indicate level of threatening processes, which can be used for assessment of anthropogenic or climate impact ranges and, thus, for setting of retention targets.

In this study RDB data for the three identified biodiversity hotspots, North Caucasus, Far East and South Siberia (see Paper 2), were digitised and put into a spreadsheet database. The database was elaborated for vascular plants, reptiles, amphibians, birds and mammals. The 288 species were described by their Russian and Latin name, Latin family and genera, category of endemism (endemics or non-endemics on the global level), category of use (hunting animals /medicinal plants or not), transboundary category (names of bordering countries), number of individuals in a hotspot (metapopulations for vascular plants) and area of distribution in a hotspot.

We propose the percentage targets (baseline and retention), reflecting differential requirements for regional protection, caused by the regional nature of the community structure and specific human threats.

In formulating conservation targets we assess the relative representativeness of rare species in hotspots by indicators and models of species richness and evenness (Venevskaia, 1996). Generally, the methodology employed in intra-hotspot comparison of RDB species representativeness follows recommendations for experimental data analyses by Maggurran, (1988) and Southwood, (1978).

We used four indicators of species richness and evenness to estimate the relative representativeness of the RDB species in the Russian hotspots:

- species richness (Margalef indicator)
- species equitability (α parameter of the logarithmic distribution for species by individual numbers)
- combination of species richness and equitability:

1. Berger-Parker indicator
2. Shennon indicator or entropy measure.

These four indicators for the RDB vascular plants and vertebrate animals were calculated to estimate relative the representativeness of the RDB species in the hotspots. Then discriminating abilities of the four indicators and the statistical significance of this discrimination were analysed and averaged values of normalised meaningful indicators were used for formulating baseline conservation targets for national scale conservation planning.

Final targets for the rare, threatened and endangered vascular plant species vary between 25% and 100% of their recorded localities. Almost all of the RDB plant species from the biodiversity hotspot in North Caucasus (76 species of the 86) require 100% of their recent distribution area in a targeted reserve network. Reserve networks in the hotspots in South Siberia and the Far East should include the entire areas of distribution for over half of the RDB plant species.

The conservation targets for the RDB animal species for the three Russian biodiversity hotspots range from 100% of the total areas in a reserve network to just a few recorded locations. Unlike the conservation targets for the RDB vascular plant species, the final targets for the animal species have similar distributions across all the three hotspots, revealing no leading role for any region in required protected areas. Indeed, between twenty and thirty species in each of the three hotspots require conservation of their entire areas of distribution, and these species comprise more than half the RDB vertebrate animal species found in the regions.

Paper 4 Venevskaia, I.,Venevsky , S., 2004. Hierarchical systematic conservation planning for a national level: example of Russia. 3. Large scale conservation plan for Russian hotspots. Biological Conservation (submitted)

Large-scale conservation plan for the three biodiversity hotspots North Caucasus, South Siberia and the Far East is suggested.

The aim of our conservation plan is the 100% achievement of identified conservation targets for the Red Data book species in one of four groups (vascular plants, amphibian/reptiles, mammals, birds) in every hotspot, while maximising the percentage of target achievement for other groups and with a minimum area requirement.

The existing statutory reserves were analysed for environmental bias and efficiency of species representation in view of the conservation targets (see Paper 3).

The high diversity of plant and animal species reflects environmental gradients (climatic and edaphic) existing in the hotspots, which are caused mainly by topographic variation. Therefore an optimal reserve network in a hotspot should represent variations in relief in order to provide a variety of habitats for biological communities and species.

However, protected areas of North Caucasus are situated in highland areas (the mean elevation for protected areas at 1007 meters is almost twice that for the entire hotspot) with moderately steep slopes (variation of elevation in the grid cells of the protected areas ranges between 235 and 483 metres in comparison with the interval 26 to 837 metres for the entire hotspot). A similar, but less profound, tendency in the topographical location of protected areas can be seen in South Siberia, where the protected areas are situated in relatively flat uplands. Protected areas of the Far East reflect both the elevation and altitudinal variation in the entire hotspot well.

Only a third (at maximum) of the conservation targets set for the Russian biodiversity hotspots is achieved within the existing reserve networks. Indeed, the percentage of fulfillment of conservation targets averaged over the four groups of species (vascular plants, birds, amphibian/reptiles and mammals) is similar for all the hotspots:

- 23% for North Caucasus;
- 34% for South Siberia;
- 19% for the Far East.

This means that additional protected areas are necessary and should be designed in view of our conservation targets.

In order to elaborate new prospective conservation areas, the data for biodiversity patterns for elaboration of prospective conservation areas in the three hotspots comprised the digitised distribution areas for 288 species from the Russian Red Book in the five taxa: terrestrial mammals, birds, amphibian, reptiles and vascular plants. The data were presented in the ARC-INFO vector format for areas of the animal species and in the point format for the locations of plant populations.

The new large-scale conservation areas within the hotspots were designed by formal optimisation procedure. The quantitative optimisation criteria for the planning protocol had two components: maximising the percentage of fulfillment of conservation targets averaged over four groups of species (vascular plants, amphibian/ reptiles, birds and mammals), and minimising area. The existing set of statutory reserves in the three Russian biodiversity hotspots was retained in our planning procedure. We incorporated the statutory reserves into the plan because considerable establishment, management and research investment has been already made. Prospective conservation areas were gradually increased, starting from the existing reserve networks, until 100% of conservation targets were achieved in one of the four groups of species (umbrella group of species).

The resulting 11 variants of prospective conservation areas in the hotspots with different umbrella groups were analysed and the best variants were suggested as the large-scale conservation plans for the regions. The largest ratio of prospective conservation area (82%) is required for North Caucasus, South Siberia follows with the 49% and less than a third of the hotspot area is proposed for the large-scale conservation plan for the Far East. The design of conservation plans for the Russian biodiversity hotspots confirmed that no single taxonomic group of species can be preferred in the elaboration of the national conservation plan. Indeed, the umbrella groups of the RDB species, providing the best conservation plans, are taxonomically different for the three hotpots.

We see this large-scale conservation plan as a starting point for further consideration by field experts and practical managers, who can conduct further selection and implementation of reserve networks within the suggested conservation areas in the three regions at a landscape level.

Conclusions

Theoretical

- Large-scale vegetation diversity can be predicted to a major extent by the climatically determined latent heat required for evaporation and the geometrical structure of landscape, described as an altitudinal difference. The climate-based model reproduces observed species numbers of vascular plants for different areas of the world with an average error of 15%
- National biodiversity hotspots can be mapped from biotic or abiotic data using the quantitative criteria for plant endemism and land use (corrected for the respective country) from the “global hotspots” approach

- Quantitative conservation targets, accounting for the difference in environmental conditions and human threats between national biodiversity hotspots, can be set using the national data for Red Data book species
- A large-scale national conservation plan reflecting the hierarchical nature of biodiversity can be designed by a combination of the abiotic method at the national scale (identification of large-scale hotspots) and the biotic method at the regional scale (analysis of species data from Red Data book)

I believe, that the majority of countries may apply the quantitative methods of hierarchical national conservation planning described in this study. Indeed, the identification of large-scale biodiversity hotspot requires climate and elevation data only, and national Red Data Books are already elaborated in many countries.

Practical

- The three biodiversity hotspots North Caucasus, South Siberia and the Far East are mapped from abiotic data. Despite the relatively small total area of the three Russian hotspots (they occupy only 3% of the entire Russian territory), these areas are inhabited by 68% of the Russian RDB species belonging to the five taxa (vascular plants, amphibian, reptiles, birds and mammals)

- A large-scale national conservation plan for Russia reflecting regional differences in biodiversity patterns and human threats is elaborated. The largest ratio of prospective conservation area (82%) is required for North Caucasus, South Siberia follows with 49% and less than a third of the hotspot area is suggested for the large-scale conservation plan for the Far East. This large scale plan can provide a basis for detailed regional conservation programmes in Russia.

Additional fine-scale data collection, application of process-oriented simulation models and the expertise of regional conservation managers may be applied to refine the large-scale conservation plan for Russia.

References

Abbitt, R.J.F., Scott, J.M.,Wilcove, D.S., 2000. The geography of vulnerability: incorporating species geography and human development patterns into conservation planning. Biological Conservation 96(2), 169-175.

Cowling, R.M., Pressey, R.L., Lombard, A.T., Desmet, P.G.,Ellis, A.G., 1999. From representation to persistence: requirements for a sustainable system of conservation areas in the species-rich mediterranean-climate desert of southern Africa. Diversity and Distributions 5, 51-57.

Cowling, R.M., Pressey, R.L., Rouget, M.,Lombard, A.T., 2003. A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa. Biological Conservation 112, 191-216.

Crisp, M.D., Laffan, S., Linder, H.P.,Monro, A., 2001. Endemism in the Australian flora. Journal of Biogeography 28, 183-198.

Foley, J.A., Costa, M.H., Delire, C., Ramankutty, N.,Snyder, P., 2003. Green Suprise? How terrestrial ecosystems could affect earth's climate. Frontiers in Ecology and the Environment 1(1), 38-44.

Fonseca, G.A.B., Balmford, A., Bibby, C., Boitani, L., Corsi, F., Brooks, T., Gascon, C., Olivieri, S., Mittermeier, R.A., Burgess, N., Dinerstein, E., Olson, D., Hannah, L., Lovett, J., Moyer, D., Rahbek, C., Stuart, S.,Williams, P.H., 2000. ...following Africa's lead in setting priorities. Nature 405, 393-394.

Francis, A.P.,Currie, D.J., 2003. A globally consistent richness - climate realtionship for angiosperms. American Naturalist 161(4), 523-536.

Laffan, S.W.,Crisp, M.D., 2003. Assessing endemism at multiple spatial scales, with an example from the Australian vascular flora. Journal of Biogeography 30(4), 511-520.

Mace, G.M., Balmford, A., Boitani, L., Cowlishaw, G., Dobson, A.P., Faith, D.P., Gaston, K.J., Humphreis, C.J., Vane-Wright, R.I.,Williams, P.H., 2000. It's time to work together and stop duplicating conservation efforts. Nature 405, 393.

Maggurran, A., 1988. Ecological Diversity and its Measurement. Croom Helm, London.

Margules, C.R.,Pressey, R.L., 2000. Systematic conservation planning. Nature 405, 243-253.

Ministry of National Resources, 2002. National Strategy for Conservation Biodiversity of Russia, Moscow.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Fonseca, G.A.B.,Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

O'Brien, E.M., 1993. Climatic gradients in woody plant species richness: towards an explanation based on analysis of southern Africa's woody flora. *J. Biogeogr.* 20, 181-198.

O'Brien, E.M., 1998. Water-energy dynamics, climate, and prediction of woody plant species richness: as interim general model. Journal of Biogeography 25, 379-398.

Pimm, S.L.,Lawton, J.H., 1998. Planning for biodiversity. Science 279, 2068-2069.

Southwood, T.R., 1978. Ecological Methods. Chapman & Hall, London.

Stolbovoi, V.,McCallum, I., 2002. CD-ROM 'Land Resources of Russia'. International Institute for Applied Systems Analysis and the Russian Academy of Science, Laxenburg.

UNEP, 1992. Convention on Biological Diversity.

Venevskaia, I.V., 1996. State of species diversity at the Northern Caucasus in comparison with other regions of Russia and some steps in biodiversity management in the region. Master of Science Thesis, CEU, Budapest.

Williams, P.H., 2000. Some properties of rarity scores for site-quality assessment. British Journal of Entomology and Natural History 13, 73-86.

Large-scale energetic and landscape factors of vegetation diversity

Sergey Venevsky, Irina Venevskaia

Abstract

We show that at large scale the species number of vascular plants can be predicted to a major extent by climatically determined latent heat for evaporation and geometrical structure of landscape, described as an altitudinal difference. Application of the energy-equivalence rule across plant communities for transpiration per area unit, and using of fractal theory for the description of habitat occupation by vegetation results in a physically-based species-energy relationship. Application of averaged global constants in the relationship results in a species-area equation with known parameters. Despite its simple form, this species-energy relationship generally reproduces global patterns of vegetation diversity for scales 10 000 and 100 000 km², and is applicable for different regions across scales from hundred to million km². The proposed theory produces more robust results compared with the correlated-based approaches, which link plant diversity patterns to climate variables, and allows the inclusion of an evolutionary component. The final climate-richness equation for vascular plants has a simple and consistent analytical form and requires easily measurable variables.

Keywords: Energy equivalence rule, fractal dimension of landscapes, scaling of vegetation diversity, species number of vascular plants

Introduction

An explanation for the striking regional differences in the distribution of species of a variety of taxons across the Earth has long been a major task of biogeography and community ecology. The number of terrestrial and fresh-water species within a sampling area of a given size has a visible latidudinal gradient, decreasing from the tropics to the poles in both hemispheres. This has been documented for morphologically different taxonomic groups (micro-organisms, trees, insects and primates) (Stevens, 1989). A second robust biodiversity pattern, originally observed first in mammalian subspecies by (Rapoport, 1982), is characterized by a positive correlation between latitude and geographical range. Around 20 phenomenological and mechanistic explanations have been suggested to describe the existing latitudinal distribution of species (Brown and Lomolino, 1998). One of the central points in debates on latitudinal gradients of species richness and geographical ranges is the role of scale dependence in mechanisms regulating biodiversity (Rosenzweig, 1995). Latitudinal patterns for different taxa are disrupted significantly by the influence of other factors like high elevation or lack of precipitation. This indicates that the latitudinal dependence of species richness should be investigated mainly as a correlate of other environmental factors, like some form of energy, available for species metabolism.

The type of available energy and the form of the relationship between this energy and the number of species in an area are some of the major hot topics in recent biodiversity debates (Whittaker, 1999; Gaston, 2000). From polar to temperate regions there is evidence, based on observations, for a positive monotonic dependence of species richness on environmental energy, although the type of energy differs between taxonomic groups (Gaston, 2000). The range of possible temporal variability in available energy appears to be a candidate for correlation with species abundance in tropics and subtropics (Gaston, 2000). Actual evapotranspiration, i.e. a measure of latent heat flux, was found to describe well tree species diversity of North America and Great Britain (Currie and Paquin, 1987); mean seasonal temperatures i.e. measures of sensible heat, were best correlates with birds and butterfly species richness in the UK (Turner et al., 1987; Turner et al., 1988). The accumulative measures of environmental energy like net primary production can also provide a good description of species richness for certain taxa (for instance for tree species richness in temperate Europe, eastern North America and East Asia (Adams and Woodward, 1989)).

On the global scale correlation - based models relating species richness for vascular plants or their subsets with potential and actual evapotranspiration, annual temperature and water deficit were found to be globally very strong (e.g. (Francis and Currie, 2003; O'Brien, 1998) at coarse spatial resolution. However, there is a need for more accurate models and theoretically sound explanations for observed correlations between climate and vegetation diversity.

Here we suggest a species-energy relationship for vascular plant species, based on the rule of energy equivalence across plant communities in the form of equivalent transpiration per area unit, which is scale transparent and takes into account seasonal distribution of available energy.

Method

Rule of energy equivalence across plant communities

The rule of energy equivalence can be derived by the more general model of energy optimization in a fractal network of branching transport tubes within an individual (Enquist et al., 1999a; Enquist et al., 1998; Enquist et al., 1999b; Enquist and Niklas, 2001). The metabolic rate of an organism is equal to its mass to the power ¾ when the energy, dissipated during transportation of essential materials (water, blood etc.), is minimized and the terminal tubes of the space-filling fractal transportation network do not vary with body size (West et al., 1997). The theory was applied successfully for the first time in plant ecology (Enquist et al., 1999a; Enquist et al., 1998; Enquist et al., 1999b; Enquist and Niklas, 2001), where previous allometric relationships were based on simple principles of Euclidian geometry. The geometric model predicts that the metabolic rate is proportional to the body mass to the power 2/3, i.e. the relationship between surface area (where the heat is lost) and the body mass (itself proportional to volume). Appling the resource distribution model through fractal networks for xylem transport of water and nutrients in vascular plants, a general model of plant vascular systems with the ¾ law analogous to animal systems was suggested (Enquist et al., 1999b). This model was justified with the 20 years time series, measuring 2283 trees of 45 species found in Costa Rica (Enquist et al., 1999a). The data fits remarkably well the ¾ power law relating the rate of gross primary production to body mass, when variation in wood density is taken into account. A –4/3 exponent for the intraspecific thinning law in plant populations was derived using this theory (Enquist et al., 1998). The new (fractal geometry based) thinning model fitted well the observed data for 251 plant species ranging from *Lemma* to *Sequoia*, i.e. 12 orders of magnitude in plant size (Enquist et al., 1998).

Two important implications of the new allocation and thinning theory can be used in plant diversity studies. First, the rate of whole-plant xylem transport or transpiration is an appropriate indices of plant metabolism, while the allometric exponents for gross photosynthesis, water and nutrient use must be equivalent due to stoichiometric constraints. Second, total energy use of plants for a given area is invariant with respect to body size. Indeed, the total transpiration rate in a plant community for a given area Q_{tot} is the product of the rate per individual Q_{ind} and the population density N. Therefore:

$$Q_{tot} = N \cdot Q_{ind} \propto \alpha M^{-3/4} \beta M^{3/4} \propto \eta M^{0}, \qquad (1)$$

where M is the averaged across the community above-ground plant biomass of an individual, α, β and η are constants, and Q_{tot} does not depend on M (Enquist et al., 1998; Enquist and Niklas, 2001).

This strongly supports the rule of energy equivalence across plant communities on a given spatial scale: despite the fact that energy is implemented in a variety of growth-form and life-history strategies, all vascular plant species attain the same optimal use of energy in a community invariant of plant size.

Species-energy relationship for vascular plants

We can include geometrical structure of landscape in the balance equation relating the available energy available to the actual energy, used for transpiration in area A:

$$\gamma * E * A = Q * A_h \qquad (2)$$

where γ is the spatially averaged ratio between transpiration and total evapotranspiration, E is the latent heat per area unit available for evapotranspiration, A_h is total amount of transpiring area for vascular plants, Q is the constant rate of transpiration in energy units per unit transpiring area, according to the hypothesis of energy equivalence across plant communities.

Actual transpiration is distributed between species, therefore:

$$\gamma * E * A = Q * \sum_{i=1}^{N_{sp}(A)} A_h^i , \qquad (3)$$

where $N_{sp}(A)$ is the number of terrestrial vascular plant species in area A and A_h^i is the total transpiring area for species i.

The total transpiration area for species i can be estimated using size-frequency distributions within an archipelago of 'self-similar' transpiring islands (vegetation patches). Given that each part of the transpiring island can be described as a linear geometric reduction of the whole with the same reduction ratios in all directions, the vegetation patches will be self-similar and their fragmentation, clumping and geometrical connectivity can be characterized by numerical characteristics used in the fractal theory (Mandelbrot, 1983).

Self-similarity of vegetation patches for large scales was observed using aerial and remote sensing images at the community level for a variety of ecosystems such as Alaskan boreal forest (van Hees, 1994), southern Texas savannah with the study area 12 million hectares (Li, 2000), mesophytic forest and cerrado ina 15 774-ha region of south-eastern Brazil (Jorge and Garcia, 1997). Self-similarity for habitat occupation by vascular plants was also found at the species level for British grasses (Kunin, 1998) and Alaskan trees (Lennon et al., 2002) from on-ground inventory using box-counting method.

(Hastings et al., 1992) suggested a power law relationship for cumulative number of self-similar islets within a distance *r* in a planar section as $N(r)=k*r^D$. The same power law was obtained by (Voss, 1988) when analyzing raster images of natural objects. He shows that the number of square pixels $N(L)$ counted within irregularly shaped islands, which can be covered at an image by sampling square *L* x *L* window, can be estimated by the power law $N(L)=k*L^D$, where *D* is a fractal dimension of an image.

On applying the power law to vegetation, the number of patches increases with the size of transpiring island *a* as a power function:

$$N^i{}_{patch}(a) = k_i * \frac{1}{h_i} * a^{\frac{D_i}{2}} , \tag{4}$$

where k_i is a constant (maximum number of patches per area), $\frac{1}{h_i}$ is the reciprocal of relative landscape lacunarity (unitless) (Milne, 1992), D_i is the fractal dimension of vegetation at a landscape for the species i . The relative local lacunarity $\frac{1}{h_i}$ is a prefactor, reflecting the influence of abiotic factors (edaphic, topographic etc.) on density and connectivity of habitats, suitable for plants (Ritchie and Olff, 1999).

(Lennon et al., 2002) analyzed fractal dimensions of species distribution for Alaskan trees and British grasses, i.e. vascular plants with different life forms and environmental conditions. They found that D_i may vary between 0.5 and 2. However, in both cases more than 70% of the total number of species possessed the range of fractal dimension between 1.3 and 1.9. The mean values of D_i for Alaskan trees and British grasses are close to each other (1.5 and 1.6 consequently). At first glance, we assume that $\frac{1}{h_i}$ and D_i depend mainly on geomorphological and long-term climatic parameters of the landscape and, therefore, relative local lacunarity and the fractal dimension of vegetation at a landscape are constant across plant species, i.e. $\frac{1}{h_i} = \frac{1}{h}$ and $D_i = D$ for the entire set of species of the area A. The total transpiring area is calculated as the sum of areas of transpiring vegetation patches, where the number of patches can be estimated using a power size-frequency distribution with $\frac{1}{h}$ and D constant across species:

$$\sum_{i=1}^{N_{sp}(A)} A_h^i = \sum_{i=1}^{N_{sp}(A)} \overline{A_i} * N^i{}_{patch}(A) = \frac{1}{h} * A^{\frac{D}{2}} * \sum_{i=1}^{N_{sp}(A)} \overline{(A_i * k_i)} = \frac{1}{h} * A^{\frac{D}{2}} * \overline{\lambda} * N_{sp}(A) \tag{5}$$

where $\overline{A_i}$ is average across species i area of a vegetation patch, $N^i{}_{patch}(A)$ is number of transpiring patches of species i in the area A, $N_{sp}(A)$ is number of vascular plant species in the area A, $\overline{\lambda} = \frac{\sum_{i=1}^{N_{sp}(A)} (\overline{A_i} * k_i)}{N_{sp}(A)}$ is the averaged constant, characterizing packing of plant vascular species in a landscape.

Combining equations (3) and (5) provides an estimate of the number of vascular plant species for an area A using general landscape characteristics and energy components:

$$N_{sp}(A) = (\frac{\gamma}{Q_{tot}^{sp}}) * E * h * A^{\frac{2-D}{2}}, \quad (6)$$

where $Q_{tot}^{sp} = Q * \overline{\lambda}$ is the average rate of transpiration for one species per area unit, constant for a certain landscape.

The maximum annual available latent heat per unit area can be obtained as a long term averaged value :

$$\overline{E} = \sum_{1}^{N_{year}} \sum_{T_j > 0} \min(LH_j; L_j) / N_{year}, \quad (7)$$

where LH_j is the portion of the radiation available for evaporation in month j with positive average temperature, L_j is energy for evaporation of available monthly precipitation in the month j, N_{year} is the length of the averaging period, which should be long enough to mask effects of inter-annual climate variability.

LH_j can be calculated (in MJ/ m²) from monthly temperature, applying the dependence of surface temperature on the radiation balance as used in the energy-balance climate models (Ramanathan and Coakley, 1978; Ramanathan et al., 1979; Balobaev, 1991):

$$LH_j = \beta * (F + K * T_j) * t_{month}, \quad (8)$$

where T_j is the mean monthly temperature (°C), $F = 49.6*10^{-6}$ MWt/m² and $K = 2.1*10^{-6}$ MWt/(m² *°C), $t_{month} = 2.592*10^6$ sec is the number of seconds in a month time, β is equal to the global value 2/3 (Baumgartner and Reichel, 1975).

$L_j = L * P_j$, where L=2.45 MJ/kg is the latent heat of evaporation, P_j is the monthly precipitation in mm (equivalent to kg/m²).

This approach allows considering seasonal distribution of available heat and water for plants, which is important for regions with summer (winter) dry climates, like India or continental Brazil.

Average global energy constants

The average global ratio between transpiration and total evapotranspiration was estimated as γ=0.52 (Choudhury et al., 1998), which results in a land surface average for the land surface transpiration of 284 mm year^{-1} or Q=696 MJ year^{-1} m^{-2} of latent heat, while annual total evapotranspiration estimated as 547 mm year^{-1} (Choudhury et al., 1998; Baumgartner and Reichel, 1975).

A rough estimate of the annual averaged rate of transpiration for one species per square meter $Q^{sp}{}_{tot}$ can be calculated from the global average number of vascular plant species per unit area, using equation (6) for species numbers of vascular plants. Indeed, when A is equal to unit area; h is set to 1; γ, Q and E have average global values, the constant $\overline{\lambda}$ can be calculated from (6) as reciprocal to the global average number of vascular plant species per unit area. We take this number 4.5 sp./m², which coincides with the average value for temperate broadleaved forest (4.38-4.8 sp./m² (Gleason, 1922)), i.e. for the zone with average for the globe climatic conditions, and calculated average value of Q^{sp}_{tot}= 154 MJ/(m² * species).

Thus we obtain an average efficiency of energy use by species $\nu = \frac{\gamma}{Q^{sp}_{tot}}$ = 0.0036 species/MJ and a species-energy relationship for vascular plants:

$$N_{sp}(A) = \nu * \sum_{1}^{N_{year}} \sum_{T_j>0} \min(\beta * (F + K * T_J) * t_{month}; L * P_j) / N_{year} * h * A^{\frac{2-D}{2}} , \qquad (9)$$

This relationship allows to estimate the number of species of vascular plant for different scales from the climate variables as long as values of the fractal dimension of vegetation fragmentation *D* and lacunarity of landscape *h* are known.

This simple species-energy relation can be developed further to include the temperature dependence of metabolic rate for plant communities in extreme climate conditions. Under such conditions the available environmental energy in the left term of equation 2 than will be limited by the metabolic rate of community and is best modeled using a negative exponent relationship with ambient temperature (Boltzman relationship, see (Allen et al., 2002)).

Average landscape parameters and species-area relationship

The lacunarity of the landscape depends on abiotic geographical features making climatic or physical barriers hindering plant species migration, like altitudinal differences, presence of water bodies and rock outcrops.

We limit our analysis on large-scale landscape features and, thus, take into account only the influence of altitudinal difference on lacunarity within area *A,* making it in a linear form:

$$h = 1 + R * \Delta H(A) , \qquad (10)$$

where *R* is the constant and $\Delta H(A)$ is the mean altitudinal difference (in m). The effect of elevation upon the surface radiation balance can be described approximately by similar expression with $R=10^{-4}$ (1/m), as higher elevations have both more open viewsheds and the solar beam travels through less air mass (Fu and Rich, 1999). We take this *R* as an initial value to describe lacunarity changes in relation to the mean altitudinal difference.

We use *D*=1.5 for the global average value of fractal dimension of landscape fragmentation, which corresponds to the classic fractional Brownian motion model of stochastic habitat occupation by plant species (Hastings et al., 1982; Sugihara and May, 1990). Similar values were obtained during an assessment of vegetation heterogeneity in New Zealand. (Nikora et al., 1999). On the North Island, vegetation patches covering areas from 1 to 10000 km² have *D*=1.42 whereas the South Island have a corresponding fractal dimension equal to 1.4.

After these considerations, energy theory logically leads us to the species-area relationship (SAR) with the coefficients, which can be directly estimated by long-term climate variables and heterogeneity of landscape:

$$N_{sp}(A) = C * (1 + R * \Delta H(A)) * A^{z} , \qquad (11)$$

where

$$C = \nu * \sum_{1}^{N_{year}} \sum_{T_j>0} \min(\beta * (F + K * T_J); L * P_j) / N_{year} \qquad (12)$$

is the average potential number of vascular plants per m² in area *A* and

$$z = \frac{2 - D}{2} \qquad (13)$$

is the floristic diversity index (Malyshev, 1975)
Setting $\Delta H(A)$ to zero in the species-energy relationship (equation 9) results in the oldest and best-documented functional dependence between number of species and area in community ecology, the Arrhenius equation (Arrhenius, 1920) with $z = \frac{2-D}{2}$=0.25,corroborated using representative field data for a wide range of animal and plant species (Preston, 1962) and several theoretical models (MacArthur and Wilson, 1967; Sugihara, 1980; Pielou, 1975). The floristic diversity index *z* has been related both to landscape fragmentation and plant species habitat occupation strategies (equation 5) and is scale dependent. (Crawley and Harral, 2001) have shown that *z* varies at small (0.1-10 m²) and intermediate (10-1000000 m²) scales from 0.2 to 0.5 and then drops at larger scales (10^8 to 10^{12} m²) to 0.2-0.3. (Malyshev, 1975) suggested ranges of *z* from 0.15 (deserts and tundra) to 0.36 (tropics). Landscapes with high vegetation fragmentation, caused for example by climate extremes like deserts or tundra, will have values of z lower than 0.25, while landscapes with uniform areas, like tropical evergreen forests, will have values close to this theoretical value, following from the classical Brownian movement model of migration (e.g. z for the five 50-ha plots of tropical forests across the globe) (see (Plotkin et al., 2000)).
With the estimated global average values for constants it is possible to calculate the number of vascular plant species in area *A* (in m²) when climate variables and altitudinal difference are known:

$$N_{sp}(A) = 3.6 * 10^{-3} * \sum_{1}^{N_{year}} \sum_{T_j > 0} \min((81. + 3.8 * T_J); 2.45 * P_j) / N_{year} * (1 + 10^{-4} * \Delta H(A)) * A^{0.25} \qquad (14)$$

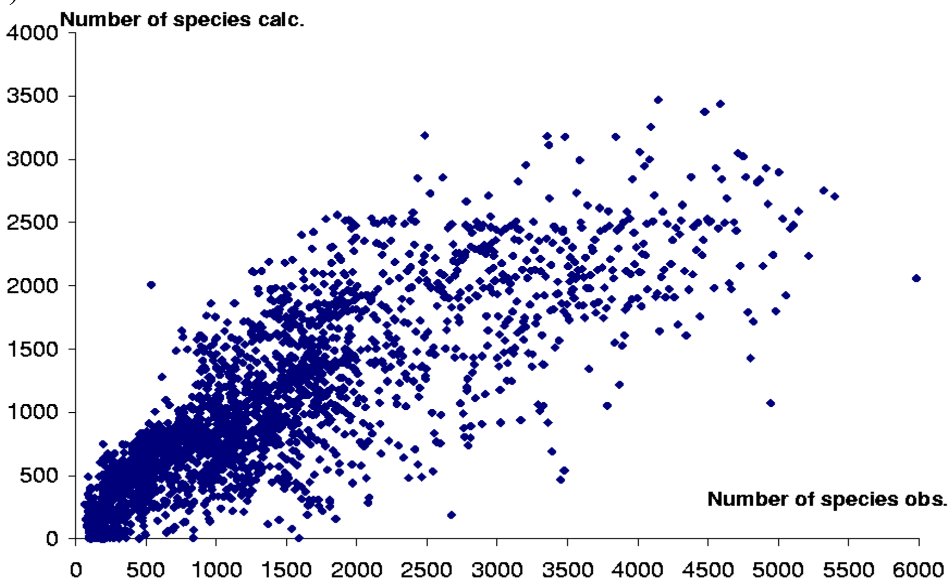
Results

To assess the applicability of the species-energy relationship (equation 14) for vascular plant species, based on average global values of fractal dimension of fragmentation and lacunarity of landscape as well as average efficiency of energy use for species, we took data on regional species abundance for a wide range of area sizes and geographical locations, and maps of global vegetation diversity. Global consistency of final climate-richness relationship in comparison with the correlation-based models (Francis and Currie, 2003) was tested over the well-known Wallace's phytogeographic provinces (Brown and Lomolino, 1998).

Global vegetation diversity

The published global maps of species number of vascular plants were assembled per sampling area of 10 000 km² (Barthlott et al., 1999), and per sampling area 100 000 km² (Malyshev, 1991) to investigate this energy-diversity relationship. The contour lines in both cases were digitized and transformed by interpolation onto a 0.5°x0.5° longitude/latitude computerized grid cell maps of observed species number (Venevskaia and Venevsky, 2002).
The maximum available latent heat in the species-energy relationship was calculated from monthly fields of mean temperature and precipitation during 1901-1998 using the CRU05 0.5°x0.5° longitude/latitude climate data, available at http://www.cru.uea.ac.uk/cru/data/. The altitudinal gradients for the sampling areas of 10000 km² and 100 000 km² were obtained for each 0.5°x0.5° longitude/latitude grid cell by averaging elevation data from the Digital Elevation Model of the World (http://edcwww.cr.usgs.gov/landdaac/gtopo30/gtopo30.html).
The correlation between the theoretical and observed species numbers of vascular plants for both global data sets are: r^2 =0.77 for the sampling area of 10000 km² and r^2=0.76 for 100000 km² (see Figure 1 a, b).

a)



b)

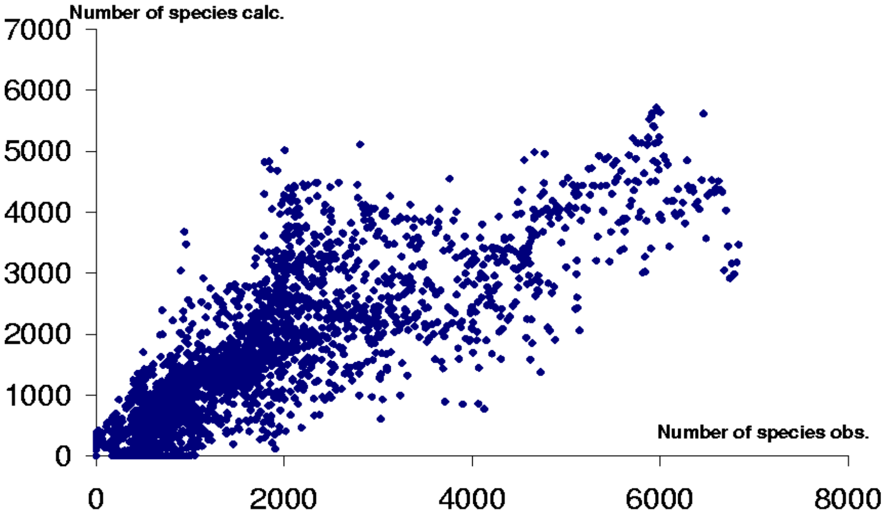


Figure 1. Observed against calculated species number of vascular plants across the globe a) per 10 000 km² b) per 100 000 km².

A step regression optimization for theoretical coefficients *ν, R* and *D* was carried out for the data set with species numbers of vascular plants per 10000 km². The correlation coefficient improved modestly (r^2=0.78) and the optimized coefficients are practically the same as theoretical, except of *R*: ν=0.0037 sp./MJ, D = 1.52 and R= 3.0 * 10^{-4} 1/m (against the theoretical value of 1.0 * 10^{-4} 1/m). This probably indicates that the altitudinal gradient affects landscape lacunarity by additional factors not only variation in the radiation balance, and, therefore, the optimized value of *R* was used in further calculations.

The absolute numbers of species and their relative spatial distribution are generally reproduced for both data sets (e.g. Figure 2 and Figure 3). Desert and tundra areas have lowest species number, tropics the highest. However, in some regions with extremely sparse vegetation (Central Australia, Arctic tundra), the numbers of species are slightly overestimated. Most likely this is the consequence of significant deviation from the global average value of landscape patchiness *D* or limitation of available energy in the form of the Boltzman's relationship for metabolic rates of community, should be considered (Allen et al., 2002). High diversity regions of South-Western Australia and

Cape Province are poorly reproduced. Accounting for specific ecological and evolutionary features of the regions may explain the bias

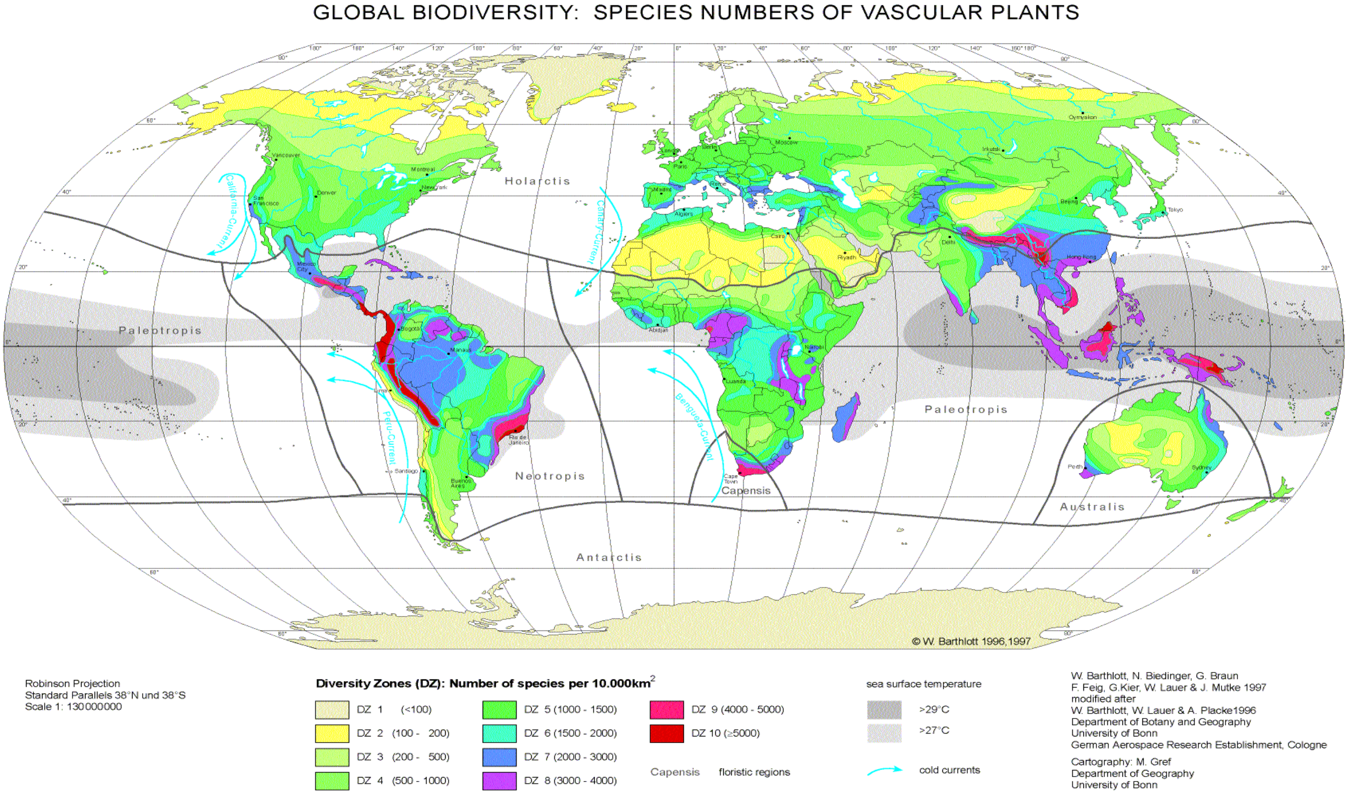


Figure 2. Observed vascular plants diversity per 10 000 km² (available on the Internet at http://www.botanik.uni-bonn.de/system/biomaps.htm)

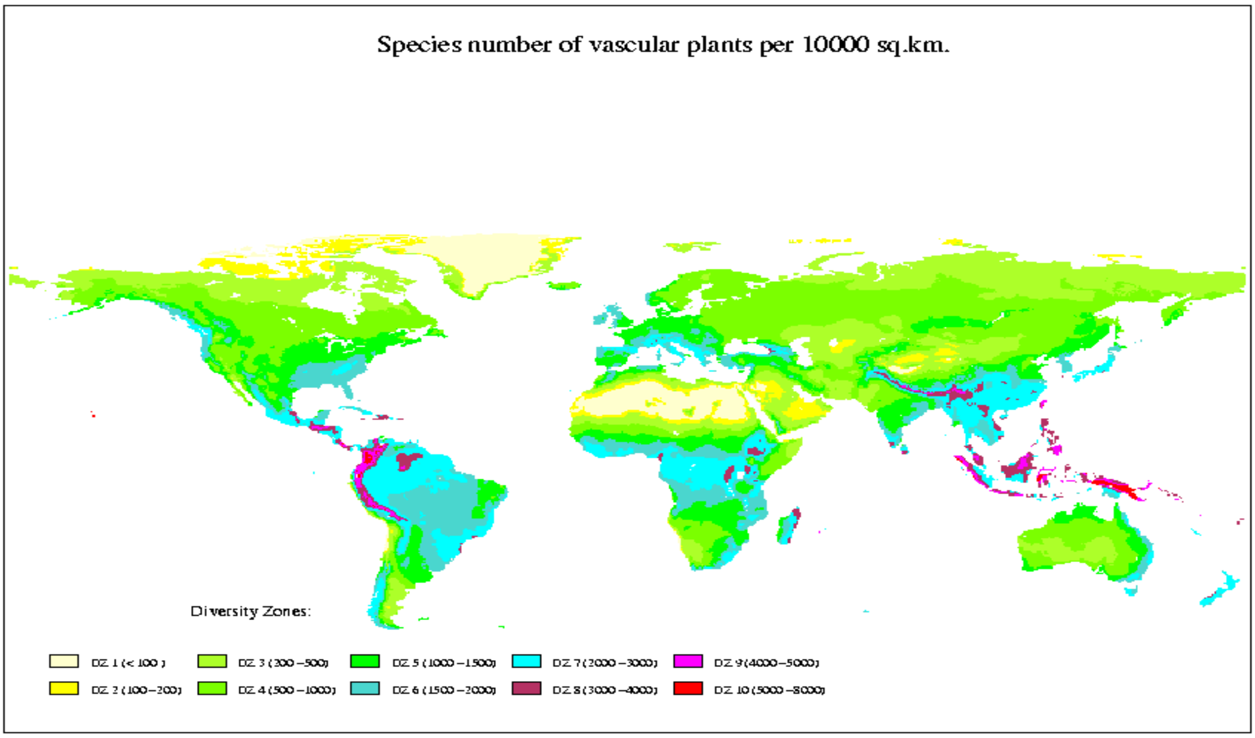


Figure 3. Simulated (using the species- energy relationship with the global average landscape and energetic parameters) vascular plants diversity pattern per 10 000 km². The 'hot spot' areas are

obtained in the Central America and the Andean, at the Brazilan Coast and in Venezuela, in the western part of Amazon basin, in the Central and Southern Africa, in Madagascar, in the Southern India, China, Indonesia and Eastern coast of Australia.

Regional vegetation diversity at different scales

The influence of scaling on model performance was estimated using the observed values for species numbers of vascular plant in the Former Soviet Union (FSU)(see (Malyshev, 1994c)), mapped over four different sampling areas: 100 km^2, 1 000 km^2, 10 000 km^2 and 100 000 km^2. The correlation coefficient between observed and simulated values for species number does not change for the three scales (0.77-0.76) and has the lowest value for the largest sampling area, i.e. 100 000 km^2 (0.6).
The mapped data for the FSU may have shortcomings for the analysis of the scale independence, because of the similarity in data processing (i.e. similar methods of spatial extrapolation) for the four different sampling resolutions. The large number of points in the global and the regional validation of the species-energy relationship also influence the correlation coefficients.
Hence, we also correlated the computed values for species number of vascular plants with the observed data from botanical surveys in different vegetation zones, from arctic tundra to the subtropics (see Table 1,2).

Zone.	Locality	Long.	Lat.	Area(km^2)	SNVPobs	SNVPcalc
Forest-tundra subzone	Nord Fugloy I. Norway	20	69	248	290	306
	Rastigassa, Lapland	26.2	69.8	265	292	322
	West Utsjoki,Finland	27.3	69.5	1075	310	409
Northern taiga subzone	Lapland Reserve,Russia	32.3	67.8	2784	523	682
	Khibin Mts, Kola Pen.	33.7	67.7	1800	429	564
Middle taiga subzone	Oulanka Nat. Park	29.2	66.4	107	429	291
Southern taiga subzone	Korpilahti, Finland	25.6	62	804	530	554
	Karku, Finland	22.1	61.3	190	503	394
	Sakyla, Finland	22.2	61	156	454	385
	Nizhne-Svirsk Reserve	33.2	60.6	410	477	474
Subtaiga subzone	Novgorod Prov.:Liubytino	33.4	58.8	700	538	598
	Latvia: mean of 4 areas	33.4	57.4	630	718	606
	Pskov Province :mean of 10 areas	29	57.3	707	700	640
	Novgorod Province.: Kholm	31.2	57.2	750	587	611
	Pskov Province.: Pushkinsky	28.9	57	750	604	622
	Latvia:mean of 6 areas	28.9	56.6	630	762	596
	Centralnolesnoy Reserve	32.9	56.5	213	546	458
	Pskov Prov.:Zhizhiza	31.4	56.3	750	607	623
	Lithuanian Natural Park	26.2	55.5	308	743	526
	Kurshskaya Kossa	21	55.1	160	630	502
	Beresinsky Reserve	28.4	54.7	760	768	663
	Naliboki, Belorussia	26.5	53.7	2400	820	909
Nemoral subzone	Belovezhs. Pucha	24	52.8	876	889	729

	Pripiatsky Reserve	28	52	603	740	665
	Shatskye Osera	23.8	51.6	710	825	702
	Polessky Reserve	28	51.5	201	602	499
	Kolbuszowa, Poland	21.8	50.2	2000	1001	952
	Strzyzow, Poland	21.6	49.8	1125	916	885
	Bieszezady Niskie	21.5	49.4	800	850	891
	Opolye Ukraine	24.5	49.3	4170	1298	1170
Forest-steppe subzone	Cherkasskiy Bor, Ukraine	31.3	49.4	417	796	593
	Olt Gorge, Romania	24.3	45.2	500	958	949
	Bucharest-Danube Plain	26.3	44.2	1597	1180	915
Genuine steppe subzone	Babadag Plateau,Romania	28.3	44.8	600	994	652

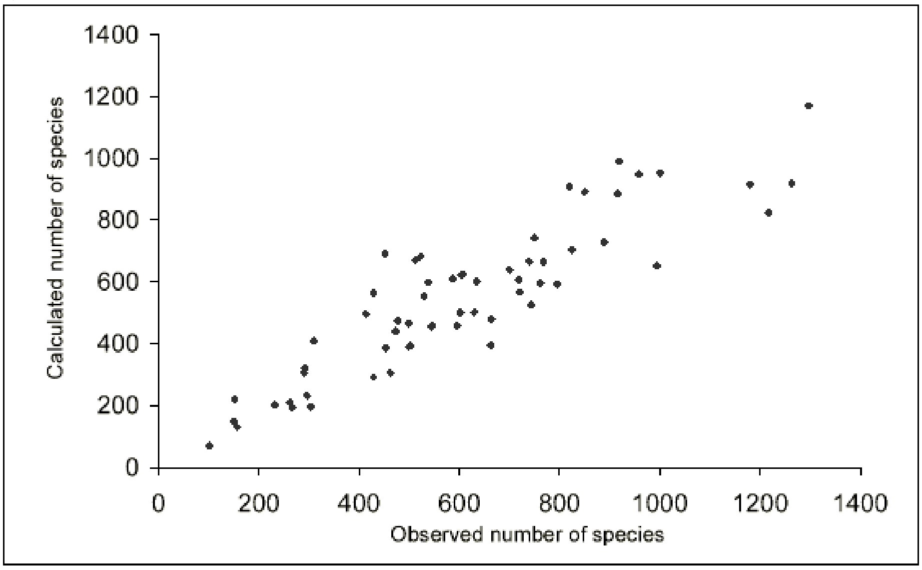
Table 1. Observed and calculated species numbers of vascular plants (SNVP) for Europe (for observed values (see Venevskaia and Venevsky, 2002)

Continent.	Locality	Long	Lat	SNVP obs	SNVP calc
Asia/Oceania	Putoran plateu	95	68	262	210
	Chukotka	-175	65	267	194
	Igarka	86.24	67.29	304	195
	Novosibirsk	83.03	56.03	463	305
	Low Amur river	135	49	500	391
	Mountains of S. Siberia	90	50	663	395
	Maritime south	131.56	43.07	473	442
	S.steppe Kazachstan	72	41	413	497
	Repetek	63.13	38.34	151	147
	Borzhomi	43.23	41.51	1263	917
	Japan Islands	140	39	544	454
	Mount Jamizo	138	35	920	990
	Savanna N. India	85	25	720	567
	Deccan peninsula	75	17	635	600
	Aden	44.5	13	232	202
Australia/New Zealand	New Zeland	168	-47	452	690
	18South East Australia	150	-36	512	670
Africa	Sahara	0	18	156	132
	Bassin of the river Kongo	15	-16	596	457
	S. Africa	28	-30	1217	824
	Rodezhia	28	-20	664	480
North America	Devon island	-90	75	102	70
	East Greenland	-45	61	152	221
	USA mainland	-100	40	500	466
	South-Eastern States	-80	35	750	742
	Alaska(Ogot.-Creek)	-150	66	297	231

Table 2. Observed and calculated species numbers of vascular plants (SNVP) for the sampling area 100 km² (Malyshev, 1991)

The data includes 59 observations from various authors for different area sizes, ranging from 100 km² to 4000 km² (Figure 4a) in Europe, Asia, Australia, Africa and North America. Despite the variety of vegetation zones, geographical locations and area sizes, the observed and calculated number of vascular plant species is strongly correlated (r^2=0.81, F=241, P<0.2*10^{-22}, slope=1.03). To investigate the upper scale limitations of the species-energy relationship based on global constants, we analyzed the survey data for countries and geographical provinces with significantly different species abundance, ranging from 0.8 to 16 million km² (Figure 4b).

a)



b)

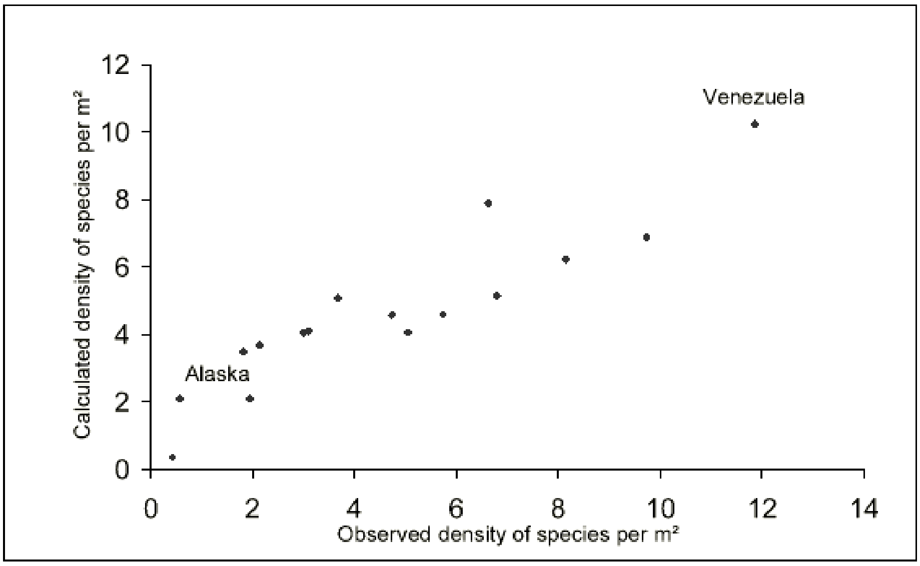


Figure 4. Observed against theoretical species numbers of vascular plants a) for various areas from 100km² to 4500 km² b) per 1 m² for large regions with variable areas from 0.84 to 16.48 million km² (observed values are collected in Heywood and Watson.L, 1995).

The correlation between averaged over the 16 regions observed and theoretical species number of vascular plants per 1 m² (r^2=0.83, F=70, P<0.0000076, slope=0.66) proves that the model can be applied for scales up to several million kilometers. However, one can see an over prediction in the regions with sparse vegetation (like Alaska) and an underestimate in regions with dense vegetation (like Venezuela). The structural differences in landscapes for the contrasting biomes, described by fractal dimension *D*, are more clearly seen for large scales, making estimates of species number more uncertain.

Testing of the climate-richness relationship for the Wallace's provinces

The climate – richness model for vascular plants, based on maximum available latent heat for evaporation and fractal characteristics of vegetation at the landscape, was compared against two recent correlation-based models (Francis and Currie, 2003) for Wallace's provinces. The (Francis and Currie, 2003) models predict angiosperm richness using non-linear regression equations combining biologically significant climate variables, like annual temperature, potential evapotranspiration (PET) and water deficit (calculated as the difference between actual evapotranspiartion and PET).

The 0.5°x0.5° longitude/latitude monthly temperature and precipitation data and the data for altitudinal gradients were rescaled by cubic approximation to 2°x2° grid cell, the spatial resolution of the correlation-based models. The coarse resolution data set was divided into six regions, representing phytogeographical provinces similar to those, used by Francis and Currie, (2003) in their study. Species richness of vascular plants in a province, predicted by the species-energy relationship, was tested against the observed SNVP from the map of Barthlott et al., (1999) at the 2°x2° spatial resolution.

The resulting correlation coefficients demonstrate higher consistency of the energy – landscape model in comparison with the correlation-based ones (see Table 3). The species-energy relationship provides robust results for all six provinces (r^2 is between 0.65 and 0.78), while the correlation in the observed/predicted family richness for the polynomial regressions from climate variables varies significantly across the Wallace's provinces, showing perfect fit for some provinces (e.g. r^2 =0.94 for Nearctic) and relatively poor performance for others (e.g. the Oriental province, where r^2 =0.42). The consistency of the energy – landscape model in different parts of the world demonstrates that the mechanisms of species and energy interactions are almost independent of the diverse evolutionary theory.

Province	Energy – landscape model r^2	Temperature models r^2	PET models r^2
Nearctic	0.75	0.94	0.94
Neotropical	0.65	0.67	0.84
Palearctic	0.77	0.79	0.89
Ethiopian	0.73	0.61	0.44
Oriental	0.71	0.43	0.42
Australian	0.78	0.66	0.78

Table 3. Performance of the three climate – richness models for vascular plants within Wallace's biogeographical provinces. Energy-landscape model: richness = maximum latent heat for evaporation x geometrical structure of landscape. Temperature models: richness = water deficit + temperature x water deficit (Francis and Currie, 2003). PET models: richness = water deficit + PET + PET^2 (Francis and Currie, 2003).

Discussion

The final climate-richness equation for vascular plants (equation 14) has a simple analytical form in comparison with formulations, suggested by the correlated-based approaches (see (O'Brien, 1993; O'Brien, 1998; Francis and Currie, 2003). These approaches predict richness by extrapolation of non-linear regression equations (Francis and Currie, 2003), which can switch input climate variables according to heuristic training rules (e.g. O'Brien, (1998)). While such models reproduce well the global diversity pattern for vascular plants at rather coarse spatial resolution (> 20 000 km²) ((Francis and Currie, 2003)), they have inevitable shortcomings, typical for regression-oriented approaches. The need to have the best fit leads to non-zero intercept in regression equations, inferring possible negative values for species richness (e.g. O'Brien, (1998)), or positive values, when the PET is absent and vascular plants can not exist (e.g. Francis and Currie, (2003)). Spatial downscaling and implementation of temporal dynamics also remain problematic for correlated-based approaches.

Despite its simple form, the proposed species-energy-relationship with average global parameters reproduces the coarse-scale global and regional patterns of vegetation diversity and works across scales from 100 to 10 000 000 km², keeping biological sense in the analytical formulation. It takes into account climate inter-annual variability and seasonality and can be used as a first estimate for species-numbers of vascular plants in theoretical and practical studies. (Stephenson, 1998) analyzed a variety of biologically meaningful climate correlates of vegetation distribution and concluded that the majority of them could not distinguish between climates with similar water and heat supplies, but different in their timing. Monthly timing of the maximum available latent heat for evaporation probably explains a better fit for the Oriental and Ethiopian phytogeographical provinces, achieved by the species-energy theory in comparison with global climate-richness regressions (Francis and Currie, 2003).

The rule of energy equivalence across plant communities, which is the basis of the suggested large-scale species – energy relationship, does not support the idea of regulation of species richness via energy induced variations in the total number of individuals in a community (Brown, 1984). Our relationship can be reformulated in a general form as: richness is equal to the average efficiency for energy use across the species pool, multiplied by the maximum available external energy flux, seasonally distributed. This product then is scaled for the area considered, according to the habitat occupation pattern at a landscape. In such form the suggested species – energy theory can be redesigned for any group of species, terrestrial or aquatic, because the largest external energy flux is usually known. The average efficiency for use of certain energy type by species within the community is the major intraspecific parameter, affecting richness in this case. For plant communities this parameter ν can be influenced either by γ the ratio between transpiration and total evapotranspiration (efficiency of energy use of the entire community) or by packing of species at a landscape $\overline{\lambda}$ (species equitability in the community).

Efficiency of energy use for a community may have different character of change with community productivity, leading to different forms of the productivity – richness relationship observed in nature, including positive, negative, hump-shaped or U-shaped (Mittelbach et al., 2001). Productivity in plant communities is determined to major extent by evolutionary adaptation to certain climate conditions and, most likely, we should expect decrease in the efficiency of energy use after the productivity optimum is reached. This may explain the predominance of the peaked form in the productivity-diversity pattern for vascular plants (Mittelbach et al., 2001), amplifying at landscape (20-200 km) and local (<20 km) scale and damping at regional (200 –4000 km) and continental scale (>4000 km) in accordance to scale variation (Crawley and Harral, 2001) of the floristic diversity index Z in the species – area relationship (equation 11).

Despite clear dominance of climate factors in global vegetation distribution, other environmental processes, like evolutionary histories, postglacial dispersal and soil nutrient dynamics may influence the recent spatial pattern. Historical factors, which played a substantial role in shaping contemporary vegetation patterns, are not described by the simplified species-energy relationship (equation 9) and may explain partly the difference between observed and simulated values for species richness.

Indeed, periodical climate changes, caused by changes in the Earth's orbit (Milankovitch oscillations), lead to considerable changes in the size and pattern of vegetation distribution. These recurrent variations in species distributions, decrease gradual speciation, increase the proportion of species formed by "abrupt" genetic mechanisms and select against specialization (Dynesius and Jansson, 2000). Generalist species with a flat plateau in the productivity – efficiency of energy use relationship are less prone to extinction, caused by high climate variability and are also not prone to gradual speciation. Thus, unevenness of species packing in a community increases and the overall richness decreases on the evolutionary time scale (see changes of parameter $\overline{\lambda}$ in the species-energy relationship (6)). When periodical climate changes on evolutionary time horizons are relatively small, species are prone to gradual speciation, favoring low dispersability and high specialization, which can be amplified by frequent disturbances and/or poor soil nutrient conditions. High equitability of species in such a community (i.e. low value of $\overline{\lambda}$) increases community efficiency of energy use ν and species richness. This can be seen for example in the Cape Floristic Region (CFR) of South Africa and southwestern Australia, the regions poorly captured by the species-energy relationship with the global constants (equation 9). The Quaternary climate was relatively stable in the CFR and high vascular plant diversity was produced recently by high speciation rates, resulting in many plants with narrow habitat specializations and small geographical ranges (Cowling et al., 1998). Similarly, shrublands and woodlands in south-western Australia persisted through the last glacial maximum (Dodson, 2001) and the regional plant richness is higher than those of south-eastern Australia, which experience also a Mediterranean climate, but where vegetation communities were assembled from refugial areas.

To investigate historical factors a next-generation model should include possible non-linear effects at the evolutionary time scale, when energy use is influenced by plant species domination/equitability in a community, regulated by associated changes in gene flow, migration rates, seeds availability and vegetation patchiness.

Vegetation patchiness *D* can deviate from its global average values, especially in the regions with intensive land use. Artificial inputs in agricultural ecosystems have significant impact upon the efficiency of energy use by species within communities with implications for species richness, which should be studied. Some plant species can be affected by rapid climate changes, resulting in competition inequities with possible loss of future diversity.

Despite evidence of self-similarity of vegetation patches at large scales for important representative ecosystems, like boreal and tropical forests (Jorge and Garcia, 1997) (Lennon et al., 2002), this is not a settled issue for the entire globe and all the terrestrial ecosystems. Most likely in real-word distributions of vascular plants many species in community may have self-similar or close to self-similar distributions, but other may not. Conditions and environmental mechanisms providing sel-similarity of vegetation patches at the species level across the globe should be identified and implemented in the next steps, while the assumption of fractal distributions for vascular plant species provides good first approximation to estimate their number.

We obtained the classic power-law SAR assuming that individual species in a vegetation community have fractal distribution with the same dimension. (Harte et al., 1999) used other anlytic methods to show that the constant fractal dimension in an assemblage of species leads to the classic SAR. However, our final SAR should be modified if a type of probability distribution for fractal dimension between species and possible variation of fractal dimension across scales are known. Different types of SAR e.g. sigmoid (Plotkin et al., 2000), or convex (Connor and McCoy, 1979), may evolve when applying different distribution types for fractal dimension, as demonstrated and discussed by Lennon et al., (2002).

Physical and biological reasons for changes of lacunarity for species distributions at a relatively large scale should be described and implemented in the final species-energy relationship. The variation of lacunarity between species and regions should be combined and related with variations in beta-diversity. Little is known about mechanisms of large scale beta-diversity so far. Condit et al., (2002) identified striking differences in beta-diversity between Central Panama and western Amazonia and argued that this fact can not be explained by limited dispersal and speciation alone. Relatively low turnover of species in western Amazonia in comparison with Panama at large distances suggests that population densities of some species may be related to not yet identified processes (Condit et al., 2002). While geological variation between plant habitats is known to increase beta-diversity (Ruokolainen and Tuomisto, 2002) other still unknown important factors influence the floras assembly in South America. Further research is needed to identify factors of beta-diversity variation at large spatial scales, relating and implementing it into general parameter as the large scale lacunarity of a region.

A next generation species-energy model based on separate parameter estimates for dominant vascular plant species in biomes (see equations 3, 5) may resolve these complicated issues. Such a model can provide a new insight for a process-based modeling study of global vegetation distribution, suggested for example by Kleidon and Mooney, (2000).

Nevertheless, a substantial fraction of the variation in vegetation diversity can be explained by limitation in available environmental energy (maximum available latent heat) and by matrix landscape structure (altitudinal difference) and estimated using global average constants.

Acknowledgments

We thank J.-H. Schellnhuber and M.Claussen for providing the financial support to this study. We acknowledge F.Badeck and two anonymous reviewers for valuable comments on this work.

References

Adams, J.M.,Woodward, F.I., 1989. Patterns in tree species richness as a test of the glacial extinction hypothesis. Nature 339, 699-701.

Allen, A.P., Brown, J.H.,Gillooly, J.F., 2002. Global biodiversity, biochemical kinetics, and the energetic-equivalence rule. Science 297, 1545-1548.

Arrhenius, O., 1920. Distribution of the species over the area. Medd. Kungl. Vetens. Nobelinst. 4(1), 1-6.

Balobaev, V.T., 1991. Geometria merzloy zony litosphery severa Azii. Nauka Press, Novosibirsk.

Barthlott, W., Biedinger, N., Braun, G., Feig, F., Kier, G.,Mutke, J., 1999. Terminological and methodological aspects of the mapping and analysis of global biodiversity. Acta Botanica Fennica 162, 103-110.

Baumgartner, A.,Reichel, E., 1975. Die Weltwasserbilanz: Niederschlag, Verdunstung and Abfluss uber Land und Meer sowie auf der Erde im Jahresdurchschnitt. R. Oldenbourg Verlag, Munchen.

Brown, J.H., 1984. On the relationship between abundance and distribution of species. American Naturalist 124, 255-279.

Brown, J.H.,Lomolino, M.V., 1998. Biogeography. 2. Sinauer, Sunderland, MA.

Choudhury, B.J., DiGirolamo, N.E., Susskind, J., Darnell, W.L., Gupta, S.K.,G., A., 1998. A biophysical process-based estimate of global land surface evaporation using sattelite and ancillary data. Journal of Hydrology 205, 186-204.

Condit, R., Pitman, N., Leigh, E.C.J., Chave, J., Terborgh, J.,Foster, R., 2002. Beta-diversity in tropical forest trees. Science 295, 666-669.

Connor, E.F.,McCoy, E.D., 1979. The statistics and biology of the species-area realtionship. The American Naturalist 113, 791-833.

Cowling, R.M., Rundel, P.M., Desmet, P.G.,Esler, K.G., 1998. Extraordinarily high regional scale plant diversity in southern African arid lands: subcontinental and global comparisons. Diversity and Distributions 4, 27-36.

Crawley, M.J.,Harral, J.E., 2001. Scale dependence in plant biodiversity. Science 291, 864-868.

Currie, D.J.,Paquin, V., 1987. Large-scale biogeographical patterns of species richness of trees. Nature 329, 326-327.

Dodson, J.R., 2001. Holocene vegetation change in Mediterranean-type climate regions of Australia. Holocene 11, 673-680.

Dynesius, M.,Jansson, R., 2000. Evolutionary consequences of changes in species' geographical distributions driven by Milankovitch climate oscillations. Proceedings of National Academy of Sciences 97(16), 91150-9120.

Enquist, B., West, G.B., Charnov, E.L.,Brown, J.H., 1999a. Allometric scaling of production and life-history variation in vascular plants. Nature 401, 907-911.

Enquist, B.J., Brown, J.H.,West, G.B., 1998. Allometric scaling of plant energetics and population density. Nature 395, 163-165.

Enquist, B.J., Brown, J.H.,West, G.B., 1999b. Plant energetics and population density. Nature 398, 573.

Enquist, B.J.,Niklas, K.J., 2001. Invariant scaling relations across tree-dominated communities. Nature 410, 655-660.

Francis, A.P.,Currie, D.J., 2003. A globally consistent richness - climate relationship for angiosperms. American Naturalist 161(4), 523-536.

Fu, P.,Rich, P.M., Design and implementation of the Solar Analyst: an ArcView extension for modeling solar radiation at landscape scales. 1999 Proceedings of the 19th Annual ESRI User Conference, San Diego, USA.

Gaston, K.J., 2000. Global patterns in biodiversity. Nature 405, 220-227.

Gleason, H.A., 1922. On the relation between species and area. Ecology 3, 158-162.

Harte, J., Kinzig, A.P.,Green, J., 1999. Self-similarity in the distribution and abundance of species. Science 284, 334-336.

Hastings, H.M., Pekelney, R., Monticciolo, R., Kannon, D.V.,Monte, D.D., 1982. Time scales, persistence and patchiness. Biosystems 15, 281-289.

Hastings, H.M., Schneider, B.S., Schreiber, M.A., Gorray, K., Maytal, G.,Maimon, J., 1992. Statistical geometry of pancreatic islets. Proceedings of Royal Society, B 250, 257-261.

Heywood, V.H.,Watson.L (Eds.), 1995. Global Biodiversity Assessment. Cambridge University Press, Cambridge.

Jorge, L.A.B.,Garcia, G.J., 1997. A study of habitat fragmentation in Southeastern Brazil using remote sensing and geographic information systems (GIS). Forest Ecology and Mangement 98, 33-47.

Kleidon, A.,Mooney, H.A., 2000. A global distribution of biodiversity inffered from climatic constraints: results from a process-based modelling study. Global Change Biology 6, 507-523.

Kunin, W.E., 1998. Exrapolating species abundance across spatial scales. Science 281, 1513-1515.

Lennon, J.J., Kunin, W.E.,Hartley, S., 2002. Fractal species distributions do not produce power-law species-area relationships. Oikos 97, 378-376.

Li, B.-L., 2000. Fractal geometry applications in description and analysis of patch patterns and patch dynamics. Ecological Modelling 132, 33-50.

MacArthur, R.H.,E.O., W., 1967. The Theory of Island Biogeography. Princeton University Press, Princeton.

Malyshev, L.I., 1975. Kolichestvenny analiz flory: prostranstvennoe raznoobrazie, uroven vidovogo bogatstva i reprezentativnost uchastkov obsledovanija. Botan. Jour. 60(11), 1537-1550.

Malyshev, L.I., 1991. Some quantitative approaches to problems of comparative floristics. In: Nimis, P.L., Crovello, T.J. (Eds.), Quantitative approaches to phytogeography. Kluwer Academic Publishers, Netherlands, pp. 15-33.

Malyshev, L.I., 1994c. Floristic richness of the USSR. In: B.A., Y. (Ed.),, Aktualnye problemy sravnitelnogo izuchenia flor. Nauka, St._Pererburg, pp. 34-87.

Mandelbrot, B.B., 1983. The fractal geometry of Nature. Freeman, San Francisco.

Milne, B.T., 1992. Spatial aggregation and neutral models in fractal landscapes. Am. Nat. 139, 32-57.

Mittelbach, G.G., Steiner, C.F., Scheiner, S.M., Gross, L.K., Reynolds, H.L., Waide, R.B., Willig, M.R., Dodson, S.I.,Gough, L., 2001. What is the observed realtionship between species richness and productivity. Ecology 82, 2381-2396.

Nikora, V.I., Pearson, C.P.,Shankar, U., 1999. Scaling properties in landscape patterns: New Zealand experience. Landscape Ecology 14, 17-33.

O'Brien, E.M., 1993. Climatic gradients in woody plant species richness: towards an explanation based on analysis of southern Africa's woody flora. *J. Biogeogr.* 20, 181-198.

O'Brien, E.M., 1998. Water-energy dynamics, climate, and prediction of woody plant species richness: as interim general model. Journal of Biogeography 25, 379-398.

Pielou, E.C., 1975. Ecological Diversity. Wiley, New York.

Plotkin, J.B., Potts, M.D., Yu, D.W., Bunyavechewin, S., Condit, R.,Foster, R., 2000. Predicting species diversity in tropical forests. Proceedings of National Academy of Sciences 97, 10850-10854.

Ramanathan, V.,Coakley, J., 1978. Climate modeling through radiative-convective models. Rev. Geophys. Space Phys. 16, 465-476.

Ramanathan, V., Lian, M.S.,Cess, R.D., 1979. Increased atmospheric CO2: zonal and seasonal estimates of the effect on the radiation energy balance and surface temperature. J. Geophys. Res. 84, 4449-4958.

Rapoport, E.H., 1982. Areography: geographic strategies of species. Pergamon, Oxford.

Ritchie, M.E.,Olff, H., 1999. Spatial scaling laws yield a synthetic theory of biodiversity. Nature 400, 557-560.

Rosenzweig, M.L., 1995. Species Diversity in Space and Time. Cambridge Univ. Press, Cambridge.

Ruokolainen, K.,Tuomisto, H., 2002. Beta-diversity in tropical forests. Science 297, 1439.

Stephenson, N.L., 1998. Actual evapotranspiration and deficit: biologically meaningful correlates of vegetation distribution across spatial scales. Journal of Biogeography 25, 855-870.

Stevens, G.C., 1989. The latitudinal gradient in geographical range: how so many species co-exist in the tropics. American Naturalist 133, 240-256.

Sugihara, G., 1980. Minimal community structure: an explanation of species abundance patterns. Amer. Nat 116, 770-787.

Sugihara, G.,May, R.M., 1990. Applications of fractals in ecology. Trends Ecol. Evol. 5, 79-86.

Turner, J.R.G., Gatehouse, C.M.,Corey, C.A., 1987. Does solar energy control organic diversity? Butterflies, moths and the British climate. Oikos 48, 195-205.

Turner, J.R.G., Lennon, J.J.,Lawrenson, J.A., 1988. British bird species distributions and energy theory. Nature 335, 539-541.

van Hees, W.W.S., 1994. A fractal model of vegetation complexity in Alaska. Landscape Ecology 9, 271-278.

Venevskaia, I.,Venevsky, S., 2002. Modelling of global vegetation diversity. 79. Potsdam Institute for Climate Impact Research, Potsdam.

Voss, R.F., 1988. Fractals in nature: from charcterization to simulation. In: Peitgen, H.-O., Saupe, D. (Eds.), The Science of Fractal Images. Springer, New York, pp. 21-70.

West, G.B., Brown, J.H.,Enquist, B.J., 1997. A general model for the origin of allometric scaling laws in biology. Science 276, 122-126.

Whittaker, R., 1999. Scaling, energetics and diversity. Nature 401, 865-866.

Hierarchical systematic conservation planning for a national level: example of Russia. 1. Identifying national biodiversity hotspots using abiotic factors

Sergey Venevsky, Irina Venevskaia[1],

Abstract

National conservation planning should operate with measures of biodiversity similar to those applied globally in order to harmonize national and international conservation strategies. Here we suggest quantitative measures which allow enable two criteria of the global biodiversity hotspots to be applied on a national level for 74 large countries, and show how these measures can be applied to map national biodiversity hotspots. The plant endemism criteria of global hotspots are captured by quantitative measures of endemism, which are approximately scale-independent and can be corrected to account for a country's environmental conditions and conservation priorities. The flexible land use criteria for national biodiversity hotspots are defined from percentage of natural vegetation remaining in the global hotspots. The minimum-area-required approach is applied to define the borders of national biodiversity hotspots using data on vascular plants species richness. We show how national biodiversity hotspots can be mapped from the species-energy relationship for vascular plants using climate, topographical and land use data when spatial pattern of species richness is not known. This methodology to map national biodiversity hotspots from abiotic factors is applied to Russia as s case study. Three Russian biodiversity hotspots, North Caucasus, South Siberia and Far East were identified. The resulting hotspots maps cover national-scale environmental gradients across Russia and although they are also identified by Russian experts, their actual geographical locations were hitherto unspecified. The large-scale national hotspots, identified for Russia, can be used for further fine-scale and more detailed conservation planning.

Keywords: National conservation planning, biodiversity hotspots, endemism, land use, species number of vascular plants.

1. Introduction

Biodiversity has aesthetic, moral, economic and environmental value for humankind (Ehrlich and Ehrlich, 1992), and therefore has to be considered as a common resource vitally important for all countries. However, management of biodiversity on a national level often does not coincide with international efforts and even on an international level a limited consensus has so far been achieved on biodiversity conservation priorities (Mace et al., 2000).

The need for a clear and practical strategy for biodiversity conservation, which is necessary to guide decision-makers on international and national levels, is widely recognized (Mace et al., 2000; Fonseca et al., 2000) and now discussed in the scientific community.

The basic concept is to elaborate and further develop the national protected area system, to satisfy both national and international conservation goals.

In our national conservation study for Russia we first identify large-scale national biodiversity hotspots from abiotic factors. We show how international methodologies of mapping global biodiversity hotspots (Myers et al., 2000) can be coherently transferred to a national level.

---

[1] Corresponding author

Endemic species of vascular plants are the basic group used to identify national biodiversity hotspots at the large scale as in the global study of (Myers et al., 2000). Several distinct ecological features make this group of species preferable over species from other taxa. Total species number of vascular plants is closely correlated with climatic energy at large scales (Francis and Currie, 2003). While climate determination of species richness is also observed for other taxa at continental and regional scales (for example for avian species richness in South America (Rahbek and Graves, 2001), the relationship between species richness and climate energy, i.e. potential and/or actual evapotranspiration, is well known only for vascular plant species (Currie and Paquin, 1987; Francis and Currie, 2003; Venevsky and Venevskaia, 2003). Regional richness hotspots demonstrate the highest overlap with rare species for vascular plants (82.6% compared with a mean value 50% for the eight taxa in South Africa (van Jaarsveld et al., 1998)). Species and endemic richness for this group coincide at the large scale, also not always in coherent way (see data of Vetaas and Grytnes, (2002) for flowering plants in Nepal and Himalayas). Another argument for the endemic vascular plant species as the basic group used to identify large biodiversity hotspots, is the visible effect of non-random spatially aggregated land conversion on extinction of native vascular plants at the regional scale (see example of California, (Seabloom et al., 2002)). While there are conflicting interests between land for human appropriation and biodiversity, observed for other taxa (e.g. amphibians, birds, butterflies, mammals, reptiles in Australia and North America, (Luck et al., 2004)), habitat heterogeneity within broad vegetation types, and not climatic energy, should be considered as the major driving force of species richness for these taxa, especially at fine scales (Kerr et al., 2001; Rahbek and Graves, 2001).

Thus, large-scale national biodiversity hotspots, identified for vascular plant species using abiotic methods, can become nuclei for further detailed conservation planning, based on fine – scale biotic data for species of other taxa.

2. Hotspots approach

2.1 Hotspots approach on a global scale

National conservation planning should operate with surrogate measures of biodiversity similar to those employed on a global scale in order to harmonize intra-national and international conservation strategies, as recommended by Convention on Biological Diversity (UNEP, 1992). Such a surrogate measure, combining groups of representative species and a level of habitat destruction, is applied within the "global biodiversity hotspot" approach (Myers et al., 2000).

Using this approach, 25 regions with exceptional species diversity and under considerable human pressure were defined as "global biodiversity hotspots" for global conservation by the Conservation International (Myers et al., 2000). The hotspots boundaries were mapped using the principle of `biological commonalities`, i.e. each area represents a separate biogeographic unit apparent either from its geographical location (like islands New Zealand, Madagascar etc) or reflecting the best-judgment of field experts (e.g. Tropical Andes and Mesoamerica) on regional flora. To qualify as a global hotspot, an area must contain at least 0.5**%** of the world's 300 000 plant species as endemics, and should have lost 70% or more of its primary vegetation (Myers et al., 2000).

2.2 Hotspots approach at regional scales

2.2.1 Hotspots at fine scales

A similar approach to 'hotspots' has been used for different taxa (birds, butterflies, lower plants), but in a more narrow sense in several regional studies (e.g. Prendergast et al., 1993; Araujo, 1999). The definition of hotspots in these studies is based on national species data (for the UK and Portugal) put in raster GIS format. A threshold for the sum of species present on a grid of 10x10 km determines whether or not a cell belongs to a richness hotspot. A threshold for rare species in a region is applied to define a rarity hotspot in an analogous way.

These studies (Prendergast et al., 1993; Araujo, 1999) provide limited support for the coincidence of richness and rarity hotspots, defined for different taxa. For example, analysis of hotspots for birds, butterflies, dragonflies, liverworts and aquatic plants resulted in two 10 km squares as hotspots for 4 taxa and 26 squares for 3 taxa from the 2761 grid cells for the UK national study (Prendergast et al., 1993). It means that identifying a number of protected areas that are representative for one or two taxa may not automatically provide conservation for other taxa.

There is some evidence, however, for cross-taxon congruence in so named "complementary areas" (Araujo, 1999; Howard et al., 1998). Complementary areas are defined as regions adding to a location underrepresented attributes of certain taxa (e.g. adding an area with some underrepresented *Genera* to a richness hotspot). For the Uganda forest reserve network it was shown (Howard et al., 1998) that areas with high complementarity for one taxon exhibited high complementarity also for all the other taxa (the study included plants, butterflies, mammals and birds). Biogeographical congruence, presented by environmentally distinguished forest types in this study, was sufficient to overcome low congruence of species richness hotspots for different taxa in the complementary areas.

2.2.2 Hotspots at large scales

Despite little evidence in cross-taxon congruence (except of complementary areas) at fine spatial resolution, there is a significant potential importance of large-scale hotspots analysis for demonstrating the effects of national-scale environmental gradients for major biodiversity areas. For example, plants, butterflies and dragonflies in the UK have a visible South-East gradient in distribution, while other groups, like bryophytes and sedges demonstrate a North-West gradient (Griffiths et al., 1999). These climatically controlled gradients are not visible at spatial scales of local hotspots, which are more easily affected by other landscape factors (edaphic, topgraphic). This results in little overlap between hotspots for different species groups (e.g. Prendergast et al., 1993).

Differences in spatial scale of analysis (970 km^2 versus 100 km^2) and its geographic extent may explain, for instance the geographical overlap among restricted-range birds and butterflies in the conterminous United States (Abbitt et al., 2000), which is not seen in the UK study (Prendergast et al., 1993).

Besides the possibility of capturing environmental gradients within a country, the large-scale hotspots analysis may help to maximize not only contemporary biodiversity patterns, but also to preserve ecological and evolutionary processes providing these patterns in the future. Indeed, representation and persistence oriented conservation planning study for the Mediterranean-climate desert in Southern Africa (Cowling et al., 1999) identified mainly large-scale spatial components (1000 to 30000 km^2) as surrogates of ecological and evolutionary processes (sand movement corridors, river catchments).

2.2.3 National hotspots

Conservation planning in a country should take into account both its area and available environmental and species distribution data. For instance, the UK species distribution data is a unique database with a spatial resolution of 10 by 10 km (Griffiths et al., 1999). It is rather unlikely that similar dataset will ever be available on a similar resolution for a country like Russia due to its large size and low accessibility in remote wilderness regions.

Lack of accurate species-range data is quite common in many countries. Therefore, we suggest a hierarchical nesting approach for biodiversity conservation in large countries, where national large-scale hotspots are first identified using abiotic factors after which existing and prospective protected areas within these hotspots can be analyzed.

3. Mapping of large-scale biodiversity hotspots on a national level

A direct application of the global hotspot criteria may be insufficient for national conservation planning due to differences in international and national conservation priorities as well as variation in political and logistical considerations at both levels. Indeed from a national point of view, species with a narrow distribution area or small populations, but which are abundant in other parts of the world, are still eligible for conservation (Gärdenfors, 2001). On the other hand, countries with extremely rich flora and fauna often can invest only limited national funds for nature conservation purposes and, therefore, have to prioritize international requirements.

Therefore, adjustment of the plant endemism and anthropogenic pressure criteria in defining global biodiversity hotspot is required to meet priorities in national conservation planning.

3.1 Adjusting of the plant endemism criteria

3.1.1 Quantitative measures of endemism for national hotspots

The global biodiversity hotspot is an area containing at least 0.5**%** of the world's plant species as endemics, which comprises a distinct biogeographical unit. Judgment of over 100 regional experts was applied to classify vascular plant species as global endemics in the 25 hotspots (Myers et al., 2000). Thus, the measure of endemism, applied in the definition of the global biodiversity hotspots, has a qualitative character. More recently, quantitative measures of endemism have been used to account for the spatial diversity of species at global and regional scales (Williams et al., 2002; Linder, 2001; Crisp et al., 2001, Laffan and Crisp, 2003). Quantitative measures of endemism were, so far, based on a regular grid cell with a defined spatial resolution. We need to elaborate quantitative measures of endemism, applicable for global and national biodiversity hotspots across different scales.

Threshold Endemism for hotspots

The Threshold Endemism measure (Gaston, 1994) defines endemism using an upper range limit. Species occupying a fixed range (as defined by experts), which can be described, for instance, by a number of grid cells (e.g. three or four grid cells) in a region are considered to be endemic. The total number of such species in each grid cell results in a spatial assessment of endemism. The definition of global biodiversity hotspot is based on a type of the Threshold Endemism (TE) measure: TE is equal to percentage of plant endemic species to the 300,000 world plant species, where endemics can only be found in this hotspot and nowhere else (Myers et al., 2000). Such a definition of TE does not account for the area of a hotspot and, thus, can be applied at any scale. Transfer of TE in a percentage form to a national level is possible only, when a prospective center of endemism or endemism area is pre-defined by experts or by theoretical considerations and number of endemic species in this area can be preliminary estimated using some formal iterative approach (see later Section 3.2).

Weighted Endemism for hotspots

While TE measures of grid cells are rather subjective and abrupt, Weighted Endemism (Williams, 2000) measures conceptually retain continuity. The Weighted Endemism (WE) measure for a grid cell is the sum of weights for species to be found in the grid cell. The weights are proportional to the inverse of species ranges in the entire region, so species with small ranges are assigned high weights, whilst species with larger ranges have progressively lower weights. The approximate WE measure can also be defined for global or national hotspots, if we assume that endemic species are those found only in a certain hotspot and not in the rest of a geographical domain. Indeed, each endemic species *i* in a hotspot has a range A_i^e which is smaller than the area of the hotspot A_{hs}, whilst the area of each non-endemic species j is less than the global or a country land area, $A_j^{ne} \le A_{total}$, so a minimum weighted endemism value for a hotspot can be estimated as:

$$WE_{\min} = \frac{S^e}{A_{hs}} + \frac{S^{ne}}{A_{total}} \le WE = \sum_1^{S^e} \frac{1}{A_i^e} + \sum_1^{S^{ne}} \frac{1}{A_j^{ne}} \quad , \tag{1}$$

where S^e is the number of endemic species, and S^{ne} the number of non-endemic species in the hotspot, respectively.

The total land area for the globe or a country is significantly larger than the area of any of the global or national hotspots. Therefore WE_{min} for a hotspot can be approximated to:

$$WE_{\min} \approx \frac{S^e}{A_{hs}} \quad . \tag{2}$$

This measure was used by ((Myers et al., 2000), Table 6) as one of the attributes, identifying the eight "hottest hotspots".

Corrected Weighted Endemism for hotspots

The corrected weighted endemism on raster basis or CWE (see Crisp et al., 2001) divides WE by the species richness of a cell, because cells with a large number of species are expected to have a high score for Weighted Endemism regardless of the actual number of endemics. The CWE for the global and national hotspots can be defined as:

$$CWE_{\min} \approx \frac{WE_{\min}}{(S^e + S^{ne})} = \frac{1}{A_{hs}} \frac{S^e}{(S^e + S^{ne})}, \tag{3}$$

which is equivalent to the relative amount of endemic species in a hotspot scaled by the reciprocal of the hotspot area.

3.1.2 Important properties of quantitative measures of plant species endemism for national hotspots

These quantitative measures of endemism can be applied for vascular plant species as the basic taxa for identification of national biodiversity hotspots (see Section 1). The TE measure of a national hotspot can be defined as the percentage of plant endemic species to the total country plant species, where endemics can be found only in this hotspot. The WE and CWE measures for national hotspots can be estimated by their minimum values (see equations 2, 3).

The threshold endemism and the weighted endemism of hotspots are linearly related in such a definition:

$$TE_{hs} = \frac{WE * A_{hs} * 100}{S^{total}}, \qquad (4)$$

where S^{total} is a total number of vascular plant species in a geographical domain (a country or the globe) and TE_{hs} is in percents.

Thus, an increase in the lower limit of TE chosen for the borders of national hotspots requires either an increase in WE of a hotspot, or an increase in area of a hotspot. Unlimited increase in area, however, distorts the rule of "biological commonality" applied for definitions of hotspots, while WE remains almost unchanged within the new wider hotspot borders. *The requirement for higher value for lower limit of TE in mapping national hotspots narrows the number of possible candidate areas for hotspots.* Both the total number of hotspots and cumulative area of the hotspots decrease, when the lower limit of TE increases (e.g. changing the TE lower limit from 0.5% to 1% in the global hotspot definition results in the exclusion of 11 hotspots from the entire set).

The CWE for the global and national hotspots in our definition is proportional to the ratio of endemics plant species in a hotspot. Representation of endemics in the entire flora of a hotspot may vary significantly. For example, the ratio of endemics to the total number of vascular plant species in the hotspot varies from 0.15 (Wallacea) to 0.81 (New Zealand) with an average value of 0.52 for the entire set of twenty five hotspots, and an average of 0.43 for the continental subset of the global hotspots (see Table 1 calculated from Myers et al., 2000).

Global biodiversity hotspot	Ratio of endemic plant species in hotspot
Wallacea	0.15
Mesoamerica	0.21
Western African Forests	0.25
Caucasus	0.25
Choc/Darien/Western Ecuador	0.26
South-Central China	0.29
Eastern Arc and Coastal Forests of Tansania	0.38
Brazil's Atalntic Forest	0.40
Succulent Karoo	0.40
Tropical Ands	0.44
Brazil's Cerrado	0.44
Western Ghats/Shri Lanka	0.45
Central Chile	0.47
California Floristic Province	0.48
Polynesia/Micronesia	0.50
Mediterranean Basin	0.52
Indo-Burma	0.52
Carribean	0.58
Sundaland	0.60
Cape Floristic Province	0.70
New Caledonia	0.76
Phillippines	0.77
SW Australia	0.80
Madagascar	0.81
New Zelandia	0.81

Table 1. Ratio of endemic plant species in the global biodiversity hotspots.

We can assume, however, that the *ratio of endemics in large-scale national hotspots will vary similarly in both range and average for countries larger than 200,000 km², as it does for the global hotspots*. (There are 74 such countries, occupying nearly 77% of the global land area). An indirect support for such an assumption can be seen from the recent analyses of changes in the corrected weighted endemism with scale in the twelve Australian centres of endemism (Laffan and Crisp, 2003). The CWE for plant species in this study was calculated on a regular grid over Australia for the spatial circular windows with diameters 1°, 3°, 5° and 7° degrees (i.e. with areas approximately equal to 8000 km², 71000 km², 196000 km² and 385000 km²). The CWE for the twelve Australian centres of endemism is approximately linearly proportional to the ratio of plant endemics to the total number of plant species, when a deviation, caused by common species, is ignored. Indeed, the spatial *CWE(x)* with circular window of diameter *x* (1°, 3°, 5° and 7° degrees) can be calculated for a hotspot as (see (Laffan and Crisp, 2003)):

$$CWE(x) = \frac{\sum_{j=1}^{S^e(i)} \frac{A_x^e(j)}{A_{tot}^e(j)} + \sum_{l=1}^{S^{ne}(i)} \frac{A_x^{ne}(l)}{A_{tot}^{ne}(l)}}{S(i)}, \qquad (5)$$

where $S^e(i)$ is the number of endemic species in a central grid cell *i*. $S^{ne}(i)$ is the number of non-endemic species in a grid cell *i*, $S(i) = S^e(i) + S^{ne}(i)$ represents the total number of vascular plant species in a grid cell *i*, $A_{tot}^e(j)$ is the total area occupied by endemic species *j* (which belongs to a grid cell *i*) over the entire Australia. $A_{tot}^{ne}(l)$ is the total area occupied by non-endemic species *l* from a grid cell *i* in the country and $A_x^e(j)$ and $A_x^{ne}(l)$ are the areas occupied by endemic and non- endemic species from a grid cell *i* in a circular window with the diameter *x*.

If diameter *x* becomes so large that all hotspots can be covered by circular windows, the areas $A_x^e(j)$ and $A_{tot}^e(j)$ become equal, by our definition of endemic species in a hotspot, and therefore:

$$CWE(x) = \frac{\sum_{j=1}^{S^e(i)} \frac{A_x^e(j)}{A_{tot}^e(j)} + \sum_{l=1}^{S^{ne}(i)} \frac{A_x^{ne}(l)}{A_{tot}^{ne}(l)}}{S(i)} = \frac{S^e}{S(i)} + \frac{S^{ne}}{S(i)} * \frac{\overline{A}_x^{ne}}{\overline{A}_{tot}^{ne}}, \qquad (6)$$

where S^e and S^{ne} are the total number of endemic and non-endemic species in a circular window, $\overline{A}_x^{ne}$ is the averaged range of non-endemic species in the circular window, $\overline{A}_{tot}^{ne}$ is the averaged range of common species in the entire country.

When the average range of common species is rather large and comparable with the country area, i.e. $\overline{A}_x^{ne} \prec \overline{A}_{tot}^{ne}$, we can assume the influence of non-endemic species on *CWE (x)* value is small and the last member of equation 6 can be neglected.

The number of species in the central grid cell of a circular window is related to the total number of species by the species-area relationship (Preston, 1962):

$$\frac{S(i)}{S(x)} = \frac{C(i)A^z}{\overline{C}(N*A)^z} = \frac{C(i)}{\overline{C}N^z}, \qquad (7)$$

where C(i) is the number of species per unit area in the central grid cell, $\overline{C}$ is the average number of species within the circular window, *A* is the area of central grid cell, *N* is the number of grid cells in a circular window, which varies from 1 to 29 in the study of Laffan and Crisp, 2003 , z is an exponent in the species-area relationship approximately equal to 0.25 (Preston, 1962).

Thus, the CWE for a window, which completely encompasses a hotspot, is approximately proportional to the ratio of endemics in the hotspot, if the influence of non-endemic species is neglected (see equations 6 and 7):

$$CWE(x) \approx (N^z * \frac{\overline{C}}{C(i)}) * \frac{S^e}{(S^e + S^{ne})} = \alpha * \frac{S_{hs}^e}{S_{hs}^e + S_{hs}^{ne}}, \qquad (8)$$

where S^{e}_{hs} and S^{ne}_{hs} are the total number of endemic and non-endemic species in a hotspot, α is a distortion factor, relating the number of species in a circular window and in a hotspot.

The CWE for the circular window of 196,000 km^2 for Australia lies in the range 0.214 – 0.636 with an average of 0.4 (see Table 1 in Laffan and Crisp, 2003). A window of 385,000 km^2 results in the CWE range 0.325 – 0.77 with an average of 0.51 (see Table 1 in Laffan and Crisp, 2003). Analysis of the CWE at multiple scales for Australia (Laffan and Crisp, 2003) has also demonstrated that rapid changes in the spatial CWE only occurs in the immediate vicinity of the twelve centres of endemism. The rate of increase in CWE with increasing window area subsequently declines.

This implies, that *at large scales we should expect a slowing in the increase of the ratio of endemic plant species in a national hotspot, when the area of this hotspot is gradually increased.*

The maximum and minimum, as well the average CWE values, at scales (analysis windows) larger than 200,000km^2 in the Australian hotspots, are comparable to the similar values for ratio of endemic plant species in the global hotspots (see Table 1 of our paper). Particularly, the global hotspot South Western Australia covering an area 310,000 km^2 has a ratio of endemics 0.80, almost equal to the CWE value of 0.77, calculated for the circular window of 385,000 km^2. Thus, the distortion factor α in equation (8) is almost equal to 1 for this national hotspot. Similarly it is approximately equal to one for North Kimberley, when we take the ratio of endemic vascular plant 0.143 for entire Northern Province of Western Australia (see Table 1 in Beard et al., 2000) and the CWE 0.162 for the circular window with the diameter 3°, which almost covers this hotspot. However more data is needed and the geographical location of a circular window may be important. We can argue that *the CWE will be approximately equal to the ratio of endemics when sizes of a circular window and a hotpot are equal.*

The areas of Australian endemism hotspots vary from 11,000 km^2 (Wet Tropics) to 310,000 km^2 (Crisp et al., 2001), while the areas of global hotspots have a broader range between 18,000 km^2 (New Caledonia) and 2 362,000 km^2 (Mediterranean Basin) (see Table 1 in Myers et al., 2000). We can assume, therefore, that *moving from the global scale to the scale of a large country (large than 200,000 km^2) will not significantly change the average CWE or the average ratio of plant endemics*, like in the case of Australia. Indeed, an almost six-fold in area of a circular window from 385,000 km^2 (maximum area in Australian case study (Laffan and Crisp, 2003) to the maximum size of global hotspot 2 362,000 km^2 may result in at most a 50% increase in CWE at maximum (see equation 8). This slow change, associated with increase of the number of grid cells in a circular window (see the first term of equation 8), can also be compensated as well by decrease in average number of species per area unit in the larger circular window (see equation 8).

3.1.3 Applicability of endemism criteria for the global hotspots on a national level

Lower limit of threshold endemism in the definition of a national hotspot

The plant endemism criteria for a national biodiversity hotspot can be defined using the TE lower limit approach as for the global hotspots: *a region in a country is defined as the national biodiversity hotspot if TE of the region is larger than a certain predefined percentage of the total number of country's plant species.* We propose transferring the recalculated TE values of the global hotspots as cut-off criteria for mapping national hotspots.

The minimum, maximum and average TE values of the global hotspots set (0.5%, 6.7% and 1.8%) can be used as a cut-off criteria in the definition of a national biodiversity hotspot, providing three possible ways for designating land for nature conservation (see 3.1.2 above).

A minimalist conservation strategy, designates biogeographic regions with the TE greater than 6.7% as hotspots, which results in the smallest number of hotspots and cumulative area of hotspots. A maximalist strategy, with the TE cut-off criteria equal to 0.5%, comprises the largest number of resulting hotspots and the largest total conservation area. The 1.8% value for the cut-off criteria TE provides a moderate strategy for defining borders of national hotspots.

Choice of strategy to define national hotspots depends on the country's economic situation and social values.

The lower limit of TE, applied for a definition of a national hotspot, should explicitly account for differences in total number of plant species, found in the country and on the globe.

Indeed, the country's total number of plant species is influenced by apparent climatic, edaphic and geological conditions. Hence, the lower limit of TE should be increased for countries with poor plant species richness and decreased in the case of rich flora in order to balance relative cumulative size of national hotspots with their relative global biodiversity value.

We suggest the following correction to the lower limit of TE in defining a national hotspot:

$$Y_L = Y_G * \frac{C_G}{C_L}, \qquad (9)$$

where Y_L is a lower limit of TE for a national biodiversity hotspot, Y_G is a lower limit of TE for a global biodiversity hotspot. C_G is the average global number of vascular plant species per unit area (or the average global carrying capacity) , C_L is the average number of vascular plant species per unit area in a country (or the average country carrying capacity). Equation (9) balances (on average) the absolute number of plant endemics per unit area in the global and the national biodiversity hotspots.

Average number of vascular plant species per unit area in a country

The average country carrying capacity for vascular plant species C_L can be estimated using national botanic surveys from the species-area relationship (SAR) (Arrehenius, 1920):

$$N_{sp}(A_L) = C_L * (A_L)^{z_L}, \qquad (10)$$

where $N_{sp}(A_L)$ is a total number of plant species in a country, A_L is an area of the country, if the average floristic diversity index z_L for the country is known.

The average floristic index for the majority of countries can be assumed to be 0.267 as a first approximation (see Appendix 1), while for 49 large countries with the area greater than 500,000 km^2 should be slightly corrected (see Appendix 1):

$$Z_L = 0.267 + 0.493 * 10^{-15} * A_L \qquad (11)$$

An alternative method is based on the species-energy relationship, which generally reproduces global patterns of vegetation diversity for scales 10,000 and 100,000 km², and is applicable for different regions across scales from hundreds to millions km² (Venevsky and Venevskaia, 2003). This climate based model reproduces the observed number of vascular plant species for different areas of the world with an average error 15% (see example for Europe in Table 1 of Venevsky and Venevskaia, 2003). Here we calculate the average potential number of vascular plants per m² in an area using the maximum annual available latent heat per m² and monthly climate variables:

$$C = \nu \sum_{1}^{N_{year}} \sum_{T_j > 0} \min(LH_J; L_J) / N_{year} = \nu \sum_{1}^{N_{year}} \sum_{T_j > 0} \min(\beta * (F + K * T_J); L * P_j) / N_{year} \qquad (12)$$

where $\nu = 0.0036$ species/MJ is the average efficiency of energy use per species, LH_j is the portion of radiation available for evaporation in month j with positive average temperature, L_j is energy for evaporation of available monthly precipitation in the month j, N_{year} is the length of the averaging period, T_j is the mean monthly temperature (°C), $F = 49.6*10^{-6}$ MWt/m² and $K = 2.1*10^{-6}$ MWt/(m² *ºC) are constants, $t_{month} = 2.592*10^6$ sec is the number of seconds in a month time, β is equal to the global value 2/3 (Baumgartner and Reichel, 1975) L=2.45 MJ/kg is the latent heat of evaporation, P_j is the monthly precipitation in mm (equivalent to kg/m²).

The species-energy relationship (12) can be used to construct geographical patterns of carrying capacity (number of species per m²) for vascular plant species on a large scale, with a spatial resolution determined by the climate data.

Thus, adjusting for the cut-off endemism criteria in the definition of a national hotspot can be done either based on a national botanic survey in appling the species-area relationship (see equations 9,10,11), or based on long-term biotic factors for a country from the theoretical species-energy relationship (see equations 9,12).

Approximation of endemism criteria by threshold for total number of vascular plant species in a national hotspot

A high concentration of endemic plants can be associated with abiotic factors, climatic, edaphic or geological history (e.g. in refuges from glaciations) or biotic factors, like mutualism. However, high endemism of vascular plants generally relates to distinct floristic provinces with high number of vascular plant species.

For example, all the 25 global hotspots spatially coincide with the areas containing high number for vascular plant species (more then 2,000 species per 10,000 km², on the global map of species numbers of vascular plants (Barthlott et al., 1999).

We can assume that areas with high endemism at a national level are also located in certain floristic provinces, which can be identified using thresholds in the total number of vascular plant species.

In this case, the lower limit of TE Y_L in the definition of a national hotspot will be substituted by the threshold for species number of vascular plants T_{snvp} :

$$T_{snvp} = \frac{Y_L * S^{total}}{\gamma} \quad . \qquad (13)$$

where S^{total} is the total species number of vascular plants (SNVP) in a country and γ is the ratio of endemics in a national hotspot. The ratio of endemics γ in any region has a value between 0.15 and 0.8 (similar to the global hotspots), which does not significantly change for this region with change of the geographical domain of study (country or globe) at the scales larger than 200,000 km^2 (see 3.1.2 above).

Therefore, as a first approximation we can adopt global average values for the ratio of plant endemics, 0.43 (the average for global continental hotspots, see Table 1) and 0.52 (the average for the global hotspots, see Table 1) as surrogates for the average ratio of plant endemics in national hotspots γ for 74 of countries which are larger than 200,000 km^2. This fixed range (0.43-0.52) will give an optimistic estimate for the ratio of plant endemics in national biodiversity hotspots. For instance, only seven from 25 global biodiversity hotspots have a ratio of endemics larger than 0.52, from which only two (South Western Australia and Cape Floristic Province) are the continental (see Table 1).

Substituting the TE lower limit with the surrogate threshold of SNVP (see equation 13) overcomes limitations in data on the abundance of plant endemics by using spatial data for vascular plant species richness only. The error, associated with this can be estimated using reciprocal to the ratio of endemics in the global hotspots (see Table 1), if we assume the similar range of variation for $\frac{1}{\gamma}$ in national hotspots (see 3.1.2). The average error for $\frac{1}{\gamma}$ is equal to 31% for the global continental hotspots only and 47% for the entire 25 hotspots. This average error should be added to the average error for an estimate of the total number of species in the country (15% for the climate predicted S^{total}) to obtain the total possible error in the threshold for species number of vascular plants T_{snvp}.

3.2 Mapping national endemism hotspots

If SNVP thresholds are defined, observed or simulated data on total species numbers of vascular plants (SNVP) for a country allows geographical location of surrogates for national endemism hotspots.

The minimum area required approach (Araujo, 1999), modified according to the structure of data (grid or raster data or just sampled data), is suggested to fulfill this task. In this approach one starts with one or several points with the richest species abundance and gradually increase the neighboring area applying some formal algorithm. The procedure stops when the total amount of the species in the area achieves a defined threshold (in our case the SNVP threshold). The minimum area approach works reliably for a grid cell formatted data for species abundance (Araujo, 1999).

3.2.1 Simulation of species number of vascular plant using abiotic factors

It is possible to estimate SNVP accurately from available latent heat for evaporation and geometrical structure of landscape, described as an altitudinal difference (Venevsky and Venevskaia, 2003). This can be done by the species-area relationship with carrying capacity *C*, calculated from the species-energy theory (see equation 12):

$$S = C * A^{0.25} * (1 + 10^{-4} * \Delta H) \ , \qquad (14)$$

where S is the species number of vascular plants in area *A* and ΔH is the maximum altitude difference in area *A* in meters.

The model calculates reliable estimates of the SNVP globally and for different regions across scales from hundreds to millions km² (Venevsky and Venevskaia, 2003), and requires only spatially distributed climate and topographic data as input. Thus, it provides the spatial pattern of SNVP for countries in a grid cell format, even when regional floristic data are not available.

3.2.2 Initial optimization points

An important problem for successful application in the minimum area approach is the choice of initial geographic objects, initial grid cells in our case, for further formal optimization. Besides the criteria of maximum possible SNVP, we should look at other attributes of these cells, namely the maximum altitude difference within a grid cell and the proximity to large water bodies to capture possible centers of endemism.

Indeed, CWE is better correlated with the standard deviation of elevation than most of the other 12 environmental variables (Crisp et al., 2001). Variation in elevation indicates availability of specialized habitats for narrow climate range of endemic plant species.

On the other hand, periodic climate changes, caused by changes in the Earth's orbit (Milankovitch oscillations), lead to considerable changes in the size and pattern of vegetation distribution at the geological time scale. These recurrent variations in species distributions decrease gradual speciation, increase the proportion of species formed by "abrupt" genetic mechanisms and select against specialization (Dynesius and Jansson, 2000). When periodical climate changes on evolutionary time horizons are relatively small, species are prone to gradual speciation, favoring low dispersability and high specialization, i.e. plant endemism, which can be amplified by frequent disturbances and/or poor soil nutrient conditions. These periodic climate changes can be damped by the influence of large water bodies, like seas or oceans, so endemism hotspots tend to be located either in coastal zones or in relative proximity to the oceans (e.g. the Australian and the global plant endemism hotspots).

The assumption, that location of recent centres of endemism is determined (in particularly) by the long-term evolution of spatial patterns of periodic climate changes gives an approximate range for distances between these centres. A distance between centres of endemism, in this case, is strongly affected by long-term quasi-stationary atmospheric conditions (or features), governing large-scale regional or global climate. These conditions, like quasi-zonal flows (subtropical and polar jet streams), mean meridional circulations (Hadley, Ferrel and polar cells), quasi-stationary atmospheric pressure systems, depressions and highs (e.g. Siberian and Azore Highs, Iceland Low) (see (Palmén and Newton, 1969) for full review), have characteristic linear scales approximately 1000 to 3000 km (Petoukhov et al., 2003).

Indeed, the distance between global centres of plant endemism, measured as the distance between two grid cells with maximum SNVP for the two nearest global hotspots, varies between 800 km (Sundaland - Wallacea) and 4,000 km (Mediterranean – Caucasus) with an average 2,200 km over all 25 hotspots. The distance between Australian centres of endemism varies between 400 km (Sydney Sandstone – Australian Alps) and 2000 km (South Western Australia – Central Australian Ranges) with an average of 950 km.

We suggest using characteristic linear scales of regional long-term quasi-stationary atmospheric conditions (i.e. their lower limit 500-1000 km and the average 1500-2000 km), to identify an initial set of points (grids) to further refine the borders of national hotspots in the 74 large countries by the minimum area approach.

3.3 Adjusting of the land use criteria

The definition of a global hotspot includes not only the endemism criteria, but also criteria for anthropogenic stress on natural ecosystems: an area should have lost 70% or more of its primary vegetation (Myers et al., 2000) to be identified as the hotspot. Definition for a hotspot on a national scale should be based on more flexible criteria, and to account for the relative intensity of land use in a country.

Indeed, together agricultural lands and urban areas cover nearly 35% of the terrestrial land (Foley et al., 2003). The percentage of land use conversion of natural vegetation in the hotspots is at least two times the global value. We use a similar criteria to identify national hotspots: an area should have lost at least two times more primary vegetation in comparison with the country's average loss as a whole.

This "twofold" land use criteria in the national hotspot definition, however, should account for the significant historical differences in regional development patterns within some countries (e.g. Alaska and the rest of USA) and treat such parts separately.

4. Mapping national biodiversity hotspots in Russia

4.1 Threshold in species number of vascular plants for Russian hotspots

4.1.1. Lower limit of threshold endemism for Russian hotspots

We calculated the lower limit of TE for Russian hotspots by two methods, using the species-area relationship (see equations 9,10,11) and from the theoretical species-energy relationship (see equations 9,10,12). Both methods produce almost identical values for a lower limit of TE Y_L, because the values of carrying capacities C_G for the globe and Russia C_L in species per unit area are similar in both algorithms:

- *Species-area relationship* The linear approximation (11) of the average floristic diversity index for Russia results in Z_L equal to 0.275, while the observed total number of vascular plant species in the country is 11,400 (Tishkov, 2002). Thus, the average number of vascular plant species per unit area in Russia C_L is to 2.62 sp/m^2 according to the species-area relationship (9). The average global carrying capacity was assumed to be 4.5 sp./m^2 (see Appendix 1)
- *Species-energy relationship* We estimated a globally distributed carrying capacity per unit area (m^2) for vascular plant species from monthly fields of mean temperature and precipitation (see equation 12) during 1901-1998 using the CRU05 0.5°x0.5° longitude/latitude climate data, available at http://www.cru.uea.ac.uk/cru/data/. Then the carrying capacity for the global area of Earth C_G and for Russia C_L were obtained by spatial averaging the 0.5°x0.5° grid data. The resulting values were, C_G = 4,22 sp/m^2 and C_L = 2,61 sp/m^2.

This comparison demonstrates that carrying capacity C_L for a large country can be feasibly estimated from climate data, even when species richness data for vascular plants is absent.

Alternatively the minimum, maximum and average TE values of the global hotspots set (0.5%, 6.7% and 1.8%) can be used as a cut-off criteria in the definition of a national biodiversity hotspot, and provides three possible ways for designating land for nature conservation (see section 3.1.2). A minimalist conservation strategy, designates biogeographic regions with the TE greater than 6.7% as hotspots, which results in the smallest number of hotspots and cumulative area. A maximalist strategy, with TE cut-off criteria at 0.5%, leads to the largest number of resulting hotspots and the largest total conservation area. Using the average TE value of 1.8% for the cut-off criteria TE provides a moderate strategy for defining borders of national hotspots.

The two alternative approaches for estimating the global and Russian carrying capacities provide ranges in the definition of the national hotspot, when we substitute minimum, average and maximum TE values of the global hotspots set (0.5%, 1.8% and 6.7%) into equation (9): the lower limit of TE should be 0.8-0.85% for the maximalist conservation strategy, 2.85 – 3% for the moderate strategy and 10.8 – 11.4%, if we want to designate the minimal area for nature protection.

We suggest the minimalist conservation strategy for Russia, which can not allocate significant national resources on nature conservation due to recent political and economic conditions. This strategy requires selection of areas in Russia, which contain 1230 – 1306 (10.8 – 11.4% from 11400) plant species as endemics.

4.1.2 Threshold for species number of vascular plants in a Russian hotspot

As maps of abundance range for each plant species in Russia are only partially completed (e.g. Malyshev et al., 1981-1997, gives a full description of Siberian vascular plants), a first attempt to identify large-scale areas with high endemism from species richness data is undertaken.

The lower limits for the number of plant endemics in Russian national hotspots (1230 – 1306 plant endemic species as the hotspot threshold) are divided by the averaged global values of γ=0.43 and γ=0.52 (see equation 13). The resulting threshold for the species number of vascular plant ranges between 2366 to 3037 species for a Russian hotspot. This threshold range for SNVP is reconcilable with the minimalist conservation strategy, which we apply. Indeed, Russian flora is relatively poor in endemics. The 0.43-0.52 average ratios for endemic plants are at their upper limit only in some ecosystems near Lake Baikal (Tishkov, 2002). Therefore, the areas of large-scale national hotspots for Russia, projected from the defined SNVP thresholds are likely to be minimal.

4.2 Minimum area approach for mapping of Russian hotspots

4.2.1 Species richness data

There are two spatially distributed data sources on SNVP for Russia, observed and simulated, for which the mapping of the national biodiversity hotspots can be carried out (Venevskaia and Venevsky, 2002):

- The maps presenting observed data illustrate the levels of species abundance of vascular plants in the sampling areas of 10^2, 10^3, 10^4, 10^5 km² for the former Soviet Union (Malyshev, 1994). These maps were constructed from 409 sites, using extra/interpolation not exceeding 10-fold size of initial area and expert evaluation of the spatial floristic diversity *z*. These data were digitised and put into raster and grid formats (0.5° x 0.5° longitude/latitude) in GIS (Arc-Info).
- The second data set was elaborated by calculating SNVP using the available latent heat for evaporation and geometrical structure of landscape, described in terms altitudinal difference (Venevsky and Venevskaia, 2003). For Russia, the maximum available latent heat in the species-energy relationship was calculated from monthly fields of mean temperature and precipitation during 1901-1998 using the CRU05 0.5°x0.5° longitude/latitude climate data, available from http://www.cru.uea.ac.uk/cru/data/. The altitudinal gradients for the different sampling areas (from 100 km² to 100,000 km²) were obtained for each 0.5°x0.5° longitude/latitude grid cell by averaging elevation data from the Digital Elevation Model (DEM) of the World (http://edcwww.cr.usgs.gov/landdaac/gtopo30/gtopo30.html). The simulated data set was also presented in grid GIS format. Both spatial data sets for Russia are similar: with correlation coefficient 0.79 (Venevskaia and Venevsky, 2002).

The purpose of this investigation was to generalize the elaborated methods to locate large-scale national hotspots in other countries.

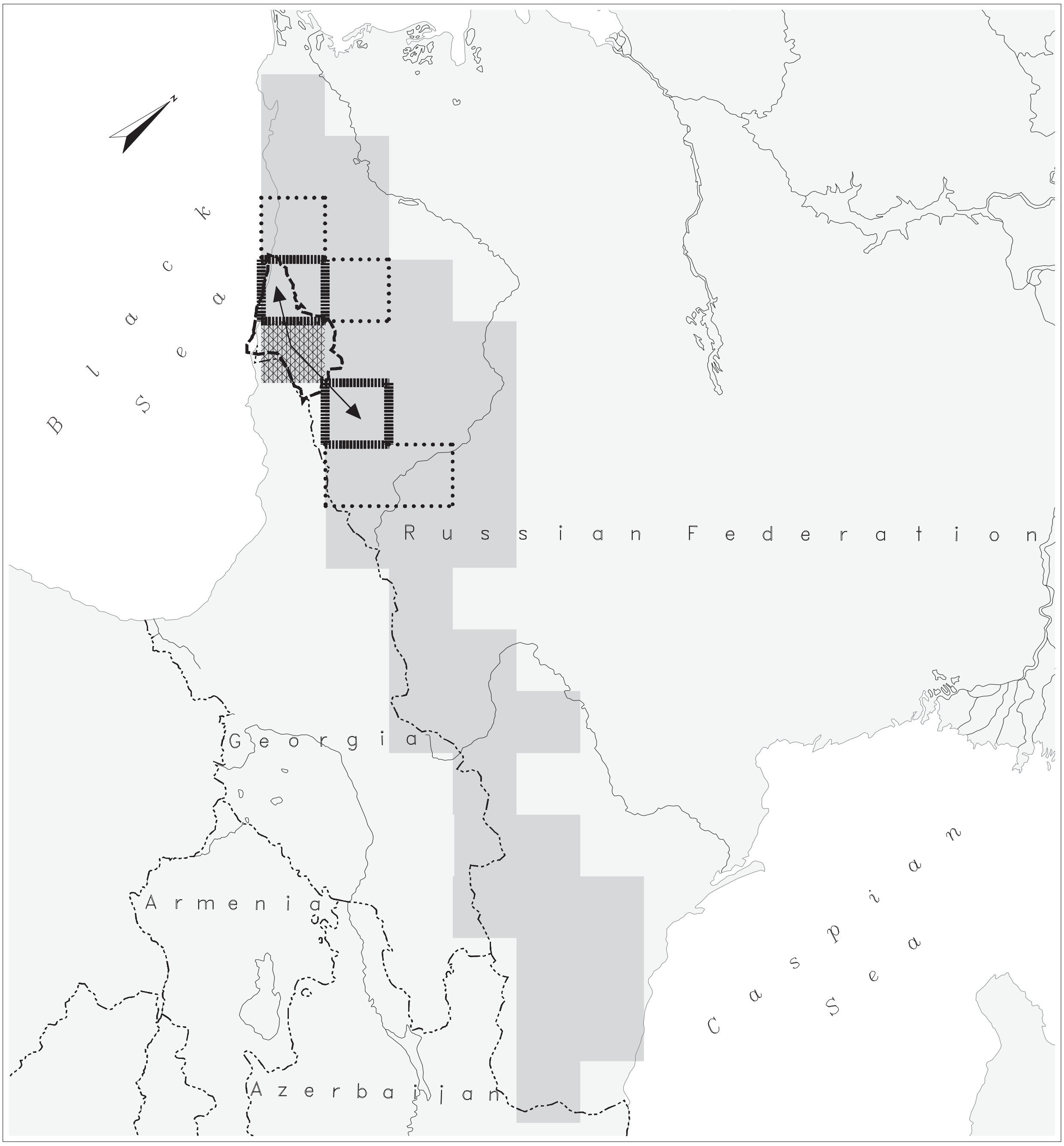
4.2.2 Optimization procedure for mapping of Russian hotspots

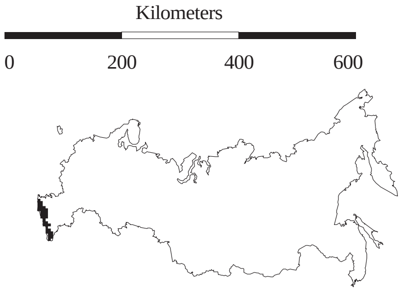
The minimum area approach was applied on the simulated species number of vascular plants for Russia with sampling area 2,500 km².

Russia’s climate is strongly influenced by the long-term quasi-stationary Siberian High with characteristic linear scale 2000-3000 km, so we set initial points a distance approximately 2000 km apart (see 3.2.2).

The four distant initial grids within the highest SNVP zones for Russia (over 1600, over 1400 and up to 1200 vascular plant species per 2,500 km² in a grid cell), with the highest altitude variation and closest to large water bodies (Black Sea, Lake Baikal, Japan Sea and Pacific Ocean) were chosen to initialize the iteration procedure (see Figure 1).

a)





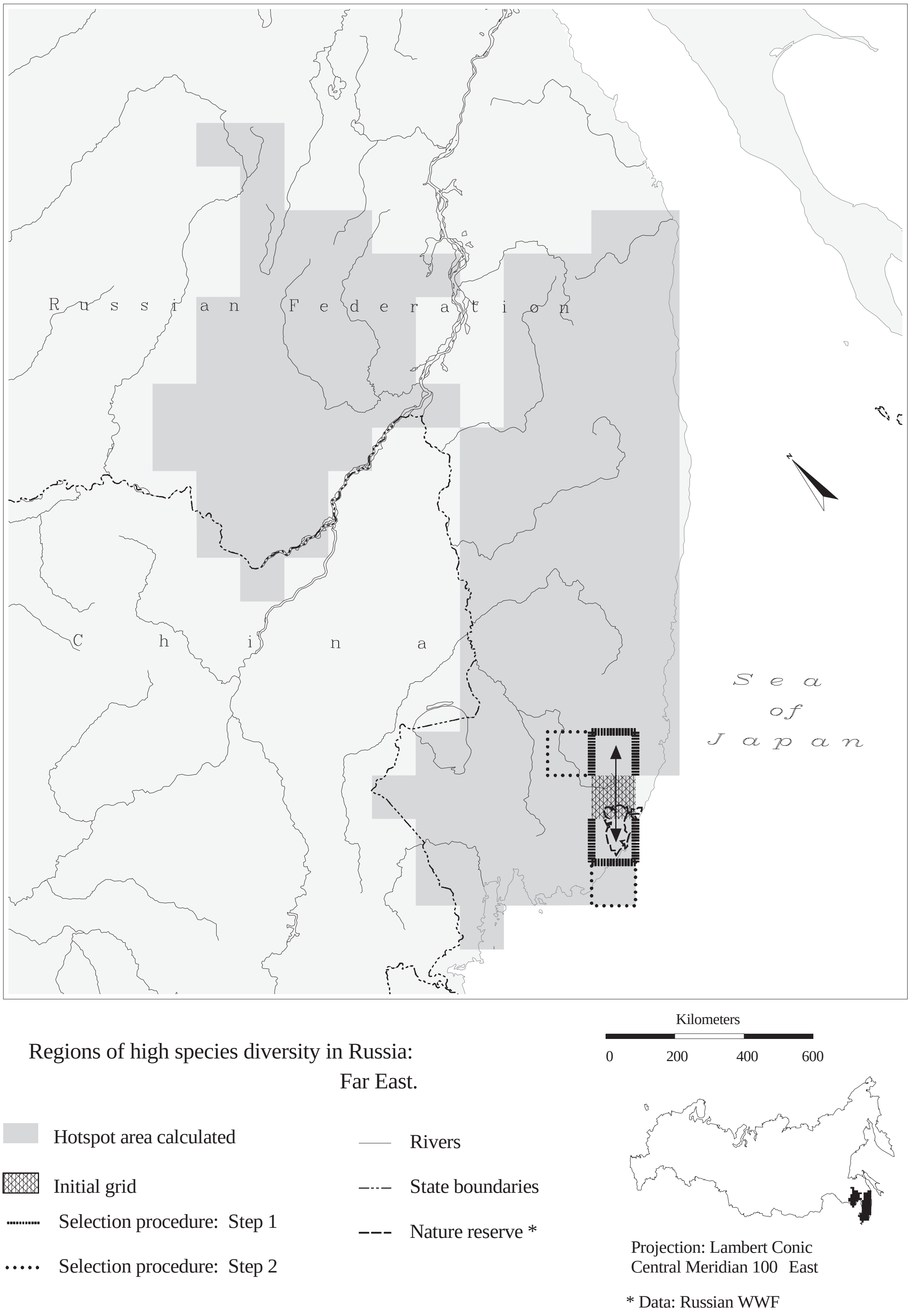
Regions of high species diversity in Russia:
North Caucasus

■ Hotspot area calculated	— Rivers
▨ Initial grid	-·- State boundaries
┅ Selection procedure: Step 1	– – – Nature reserve *
····· Selection procedure: Step 2	

Projection: Lambert Conic
Central Meridian 100 East

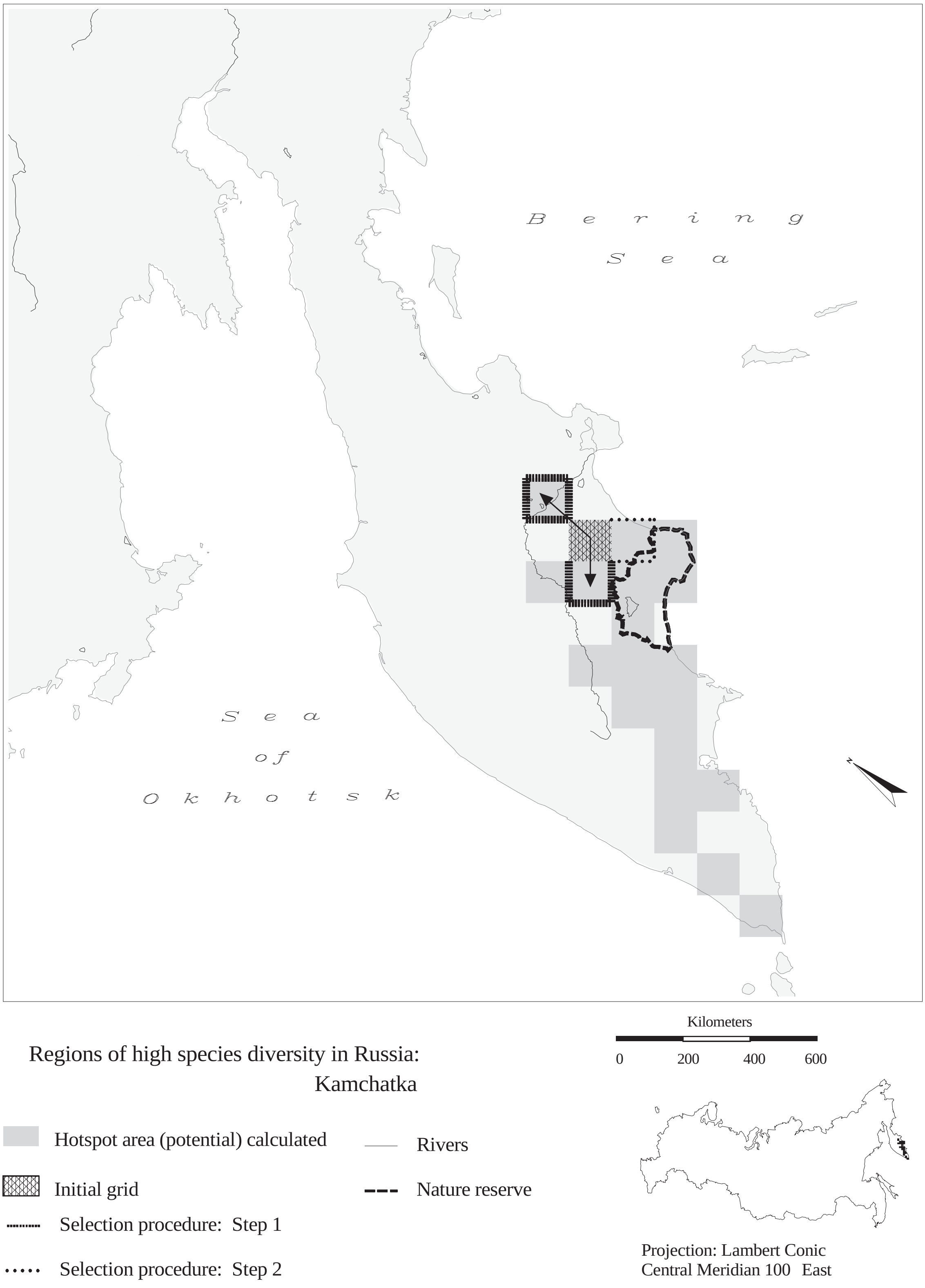
* Data: Russian WWF

b)



Regions of high species diversity in Russia: Far East.

c)



Regions of high species diversity in Russia:
Kamchatka

Projection: Lambert Conic
Central Meridian 100 East

* Data: Russian WWF

d)

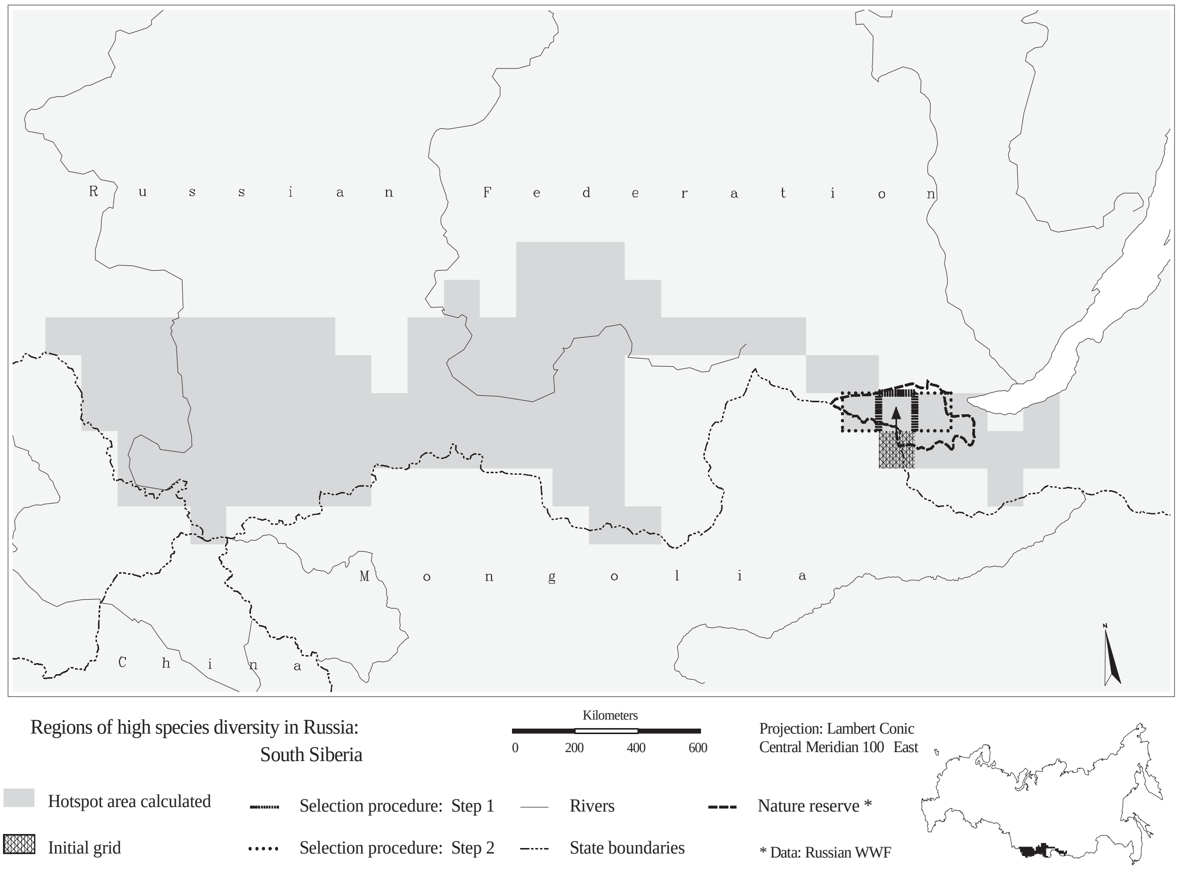


Figure 1. Optimization procedure to identify Russian biodiversity hotspots. Four distant grid cells were identified within 1) highest SNVP zones; 2) near the large water bodies; 3) with highest altitude difference: a) 39.5º E, 43.5º N near city Sotchi in Kavkazsky biosphere natural reserve; b) 135º E, 44º N near mountain Oblatchnaia in Lazovsky nature reserve; c) 161º E, 56º N near mountain Klutchevskaia Sopka and Kronotsky biosphere nature reserve; d) 102º E, 51º N near the national border in Tunkinsky national park. Grid cells were gradually included, from the neighboring vegetation diversity zones following the smallest gradient in SNVP change (see arrows), until the threshold criteria was reached.

Despite the fact that the three criteria for selection of initial grids were based only on abiotic factors (SNVP was calculated from climate and altitude difference), grid location coincides well with the existing nature reserves, designed using biological methods (see Figure 1).

Regions (encircled four initial grids) were increased iteratively by including the grid cells within the next "+/- 50 species" isoclines following paths with minimal absolute gradients in SNVP value (see Figure 1). The procedure was stopped when species number of vascular plants in the three regions reached a threshold value between 2366 and 3037 species. The iterative procedure for Kamchatka resulted in the 1772 vascular plant species and was stopped when the gradient in SNVP fell to zero. The SNVP value for identified area in Kamchatka is at the lower error limit for threshold species number of vascular plants T_{snvp} (see Section 3.1.3), so the region can be only considered a potential national biodiversity hotspot.

4.3 Resulting biodiversity hotspots for Russia

The biodiversity hotspots for Russia in the Northern Caucasus, South Siberia and Far East, comprise approximately 3% of the entire country area, were obtained by the iterative procedure (see Table 2 and Figure 2).

	Northern Caucasus	South Siberia	Far East
Area (km²)	68761	182875	229691
Number of vascular plant species	2501	2845	2641

Table 2. Three Russian biodiversity hotspots

The experts also identified these regions as the most important zones with the highest floristic and faunistic diversity in Russia (Ministry of National Resources, 2002), but they did not specify their actual areas.

Environmental gradients across Russia are well represented in the large-scale hotspots. These areas are characterized by changes in climatic conditions from South to North and high landscape variation from mountainous to plain terrain. These gradients are apparent in the high diversity of vegetation zones included in the national hotspots (see Table 3).

Russian hotspots	Vegetation zones	Number of zones
Northern Caucasus	Temperate forest, Steppe, Semideseret	3
South Siberia	Temperate forest, Steppe, Semideseret, Middle taiga, Southern taiga	5
Far East	Temperate forest, Steppe, Pre-tundra, Middle taiga, Southern taiga	5

Table 3. Vegetation zones in the hotspots

Six vegetation zones from a total of eight, as identified for Russia by Stolbovoi and McCallum (2002), are represented in the three biodiversity hotspots, excluding only tundra and northern taiga.

These hotspots comprise almost completely regional floras distinguished by geobotanists and, thus, fit to the principle of "biological commonalities", applied in the global hotspots approach. The Far Eastern large-scale hotspot includes most of the Ussurean and Burean floras (Qian et al., 2003), the South-Siberian hotspot contains several regional floras, including floras of the southern coast of the Lake Baikal, high-mountains of the Eastern Sayan and the Central Altai (Malyshev et al., 1981-1997). The Northern Caucasian hotspot includes the elements of Caucasian flora (Dolukhanov, 1966) within Russia's borders.

Figure 2. Simulated Russian biodiversity hotspots.

The three hotspots identified for Russia have not lost 70% of their primary vegetation, and, thus, are not recognized by the land use criteria for the global hotspots: 55% of the area in the Northern Caucasus without land use, South Siberia and Far East have 92% and 90% of their natural vegetation, respectively, according to the global dataset on contemporary land use (Ramankutty and Foley, 1998).

However, for Russia such anthropogenic impact should be considered as relatively high. Indeed, two identified areas are situated in the Asian part of Russia, which is almost unpopulated. The ratio of agricultural lands to total area is largest for South Siberia and Far East in the territory east of the Ural Mountains. In addition, these areas contain a high concentration of forest industry enterprises and achieve the highest forest harvest in Russia. There are fifty large saw mills in the Far East area and eighteen in South Siberia (Korovin et al., 1998). The annual harvest estimated at 15,548 thousand cubic meters of wood for Far East and 8352 thousand cubic meters for South Siberia is the highest for Russian economic regions (Korovin et al., 1998).

The hotspot in the Northern Caucasus is mainly mountainous terrain, with low suitability for agriculture. Nevertheless, almost half of this area is actively used for economic purposes, which is not observed in any other mountainous region of Russia (Ramankutty and Foley, 1998). Here land use includes intensive grain and fruit production, land for pasture, machinery, cement and oil refining industry and coastal tourism in the western part.

The lands occupied by agriculture and urban settlements comprise nearly 9% of the total area of Russia (Ramankutty and Foley, 1998). However, these lands are distributed very unevenly across Russia. Twenty three percent of primary vegetation has been lost in the European part of Russia to the West of 60ºE (approximately Ural Mountains range), and only four percent in land use in the remaining Asian part.

Within the broad subdivision of Russia into European and Asian (necessary not only because of the large area of the country, but also due to historical patterns of land use in Russia), all the identified hotspots exceed the suggested "twofold anthropogenic stress" criteria (see 3.3).

Indeed, the Northern Caucasus biodiversity hotspot has lost 45% of its primary vegetation, i.e. approximately double than the twenty three percent of land conversion, across the European part of Russia. The two Asian national hotspots also have two times the land use conversion rates compare with the entire Asian territory of Russia.

5. Discussion

A suggested quantitative approach to transfer the endemism and land use threshold criteria from the global to a national level with further mapping of national biodiversity hotspots can be performed in any of the seventy four largest countries. The approach does not rely on data on species ranges rather the global spatially distributed data sets for climate, land use and topography which are generally available.

Mapping national biodiversity hotspots allows implementation of a hierarchical nesting approach for biodiversity conservation in large countries. Large-scale regions suitable for species persistence and retention, but which are exposed to anthropogenic pressure, are first identified using only abiotic factors. Existing and prospective protected areas can then be analyzed using fine-scale species abundance data for different taxa within these hotspots.

The species-energy theory for vascular plants (Venevsky and Venevskaia, 2003) provides reasonable estimates of the geographical pattern for species number of vascular plants, necessary for identification of species richness hotpots. However, the ratio of endemics in hotspots may depend on historical evolutionary factors and, thus, remains the most uncertain characteristic in the described methodology. A constant ratio between species richness and species endemism for vascular plants can be used to identify large scale national hotspots as a first approximation, when abiotic factors affecting distribution of narrow range species are not studied. Indeed, areas with high species richness may also contain a high number of endemic species, but not always in a coherent pattern (Whittaker et al., 2002). For example, Vetaas and Grytnes, (2002) showed that at a large scale, species richness and endemic richness coincide in Nepal and the Himalayas. However, the interval of maximum species richness (1500 m –2500 m) is below the interval of maximum species endemism (3800 m – 4200 m). This may relate to a hard boundary, (the equilibrium line at *c.* 4000 m of the last glacial maximum), which caused an increase in the extinction rate above 4000 m and enhanced the probability of isolation with further speciation of neoendemics. A next generation model of plant vascular species diversity should include geological and historical factors (insularity, glaciations) for a better description of endemism.

It is important to compare the large-scale national biodiversity hotspots identified by experts using biotic data with those predicted from abiotic factors in our empirical approach or future more sophisticated theories of vascular plant endemism. Such a study may facilitate both abiotic and biotic methods for detection of large scale biodiversity hotspots.

Mapping national hotspots of plant endemism allows us to distinguish areas where limits of anthropogenic pressure has not yet reached be a cause for concern in the future. Land-use change was identified as the most critical driver in biodiversity loss (Sala et al., 2000). Therefore, investigation of future anthropogenic pressure within a set of land use scenarios is necessary in order to define critical time periods, when adaptation and mitigation measures should be undertaken to preserve biodiversity in the identified plant endemism hotspots.

Recent studies on extinction risk from future climate change (Thomas et al., 2004) and the recognized role of climate change in the biggest mass extinction in the past, the end-Permian event (Benton and Twitchett, 2003), reveal a necessity to implement of new criteria in the definition of global and national biodiversity hotspot. Such a criteria should somehow reflect the possible rate of future climate induced changes in biodiversity, showing how "hot" a hotspot will become.

Acknowledgments

We acknowledge Dieter Wallschlaeger and Franz Badeck for providing an opportunity to conduct this study and Stephen Sitch for English editing. We appreciate constructive and valuable comments and recommendations of two anonymous reviewers.

APPENDIX 1. Definition of floristic diversity index in the Arrehenius equation for a country

The floristic diversity index *z* has been related both to landscape fragmentation and plant species habitat occupation strategies and is scale dependent. Crawley and Harral, (2001) have shown that *z* varies at small (0.1-10 m^2) and intermediate (10-1000000 m^2) scales from 0.2 to 0.5 and then drops at larger scales (10^8 to 10^{12} m^2) to 0.2-0.3. Malyshev, (1975) suggested ranges of *z* from 0.15 (deserts and tundra) to 0.36 (tropics). Z=0.25 was obtained using representative field data for a wide range of animal and plant species (Preston, 1962) and several theoretical models (MacArthur and Wilson, 1967; Sugihara and May, 1990; Pielou, 1975). It can, at first glance, substitute an average floristic diversity index for a country. At large scales, however, there are data where z is greater than 0.25 (Rosenzweig, 1995) and this deviation becomes significant for the entire land area of the Earth. Indeed, if we take the average global carrying capacity to be 4.5 sp./m^2, which coincides with the average value for the zone with average for the globe climatic conditions (temperate broadleaved forest 4.38-4.8 sp./m^2 (Gleason, 1922), we get the average floristic diversity index $Z = 0.34$ for the globe from the Arrehenius equation.

The land area of countries varies by seven orders of magnitude from 0.2 km^2 (Bassas da India) to 16,995,800 km^2 (Russia) with an average value 522,470 km^2. Thus, an average floristic diversity index for a country Z_L should be corrected for area. From 251 countries and dependencies of the world (http://www.ed-u.com/area-of-countries-a.htm) 63 have an area less than 500 km^2; these are small islands or city-states, which we do not consider in our analysis, while the 49 largest countries are larger than 500,000 km^2.

The majority of countries (a hundred thirty nine) have an area between 500 and 500,000 km^2 for which we apply $Z_L = 0.267$. This value was obtained from an analysis for vascular plant species at a range of spatial scales in East Berkshire (465 km^2), Berkshire as a whole (1876 km^2) and Great Britain (229900 km^2) as the slope of regression of log(mean species richness) against log(area) (Crawley and Harral, 2001). A linear approximation for Z_L ranging from 0.267 to 0.34 in an area 500,000 km^2 to 148,300,000 km^2 (the entire global land area) can be assumed to estimate an average floristic diversity index in the 49 large countries:

$$Z_L = 0.267 + 0.493 * 10^{-15} * A_L$$

References

Abbitt, R.J.F., Scott, J.M.,Wilcove, D.S., 2000. The geography of vulnerability: incorporating species geography and human development patterns into conservation planning. Biological Conservation 96(2), 169-175.

Araujo, M.B., 1999. Distribution patterns of biodiversity and the design of a representative reserve network in Portugal. Diversity and Distributions 5, 151-163.

Arrehenius, O., 1920. Distribution of the species over the area. Medd. Kungl. Vetens. Nobelinst. 4(1), 1-6.

Barthlott, W., Biedinger, N., Braun, G., Feig, F., Kier, G.,Mutke, J., 1999. Terminological and methodological aspects of the mapping and analysis of global biodiversity. Acta Botanica Fennica 162, 103-110.

Baumgartner, A.,Reichel, E., 1975. Die Weltwasserbilanz: Niederschlag, Verdunstung and Abfluss uber Land und Meer sowie auf der Erde im Jahresdurchschnitt. R. Oldenbourg Verlag, Munchen.

Beard, J.S., Chapman, A.R.,Goia, P., 2000. Species richness and endemism in the Western Australian flora. Journal of Biogeography 27, 1257-1268.

Benton, M.J.,Twitchett, R.J., 2003. How to kill (almost) all life: the end Permian extinction event. Trends in Ecology and Evolution 18(7), 2003.

Cowling, R.M., Pressey, R.L., Lombard, A.T., Desmet, P.G.,Ellis, A.G., 1999. From representation to persistence: requirements for a sustainable system of conservation areas in the species-rich mediterranean-climate desert of southern Africa. Diversity and Distributions 5, 51-57.

Crawley, M.J.,Harral, J.E., 2001. Scale dependence in plant biodiversity. Science 291, 864-868.

Crisp, M.D., Laffan, S., Linder, H.P.,Monro, A., 2001. Endemism in the Australian flora. Journal of Biogeography 28, 183-198.

Currie, D.J.,Paquin, V., 1987. Large-scale biogeographical patterns of species richness of trees. Nature 329, 326-327.

Dolukhanov, A.G., 1966. Vegetation cover. In: I.P., G. (Ed.),, The Caucasus. Nauka, Moscow, pp. 223-255.

Dynesius, M.,Jansson, R., 2000. Evolutionary consequences of changes in species' geographical distributions driven by Milankovitch climate oscillations. Proceedings of the National Academy of Sciences of the United States of America 97, 9115-9120.

Ehrlich, P.A.,Ehrlich, A.H., 1992. The value of biodiversity. Ambio 21(3), 219-226.

Foley, J.A., Costa, M.H., Delire, C., Ramankutty, N.,Snyder, P., 2003. Green Suprise? How terrestrial ecosystems could affect earth's climate. Frontiers in Ecology and the Environment 1(1), 38-44.

Fonseca, G.A.B., Balmford, A., Bibby, C., Boitani, L., Corsi, F., Brooks, T., Gascon, C., Olivieri, S., Mittermeier, R.A., Burgess, N., Dinerstein, E., Olson, D., Hannah, L., Lovett, J., Moyer, D., Rahbek, C., Stuart, S.,Williams, P.H., 2000. ...following Africa's lead in setting priorities. Nature 405, 393-394.

Francis, A.P.,Currie, D.J., 2003. A globally consistent richness - climate relationship for angiosperms. American Naturalist 161(4), 523-536.

Gärdenfors, U., 2001. Classifying threatened species at national versus global levels. Trends in Ecology and Evolution 16(9), 511-516.

Gaston, K.J., 1994. Rarity. Chapman & Hall, London.

Gleason, H.A., 1922. On the relation between species and area. Ecology 3, 158-162.

Griffiths, G.H., Eversham, B.C.,Roy, B.D., 1999. Integrating species and habitat data for nature conservation in Great Britain: data sources and methods. Global Ecology and Biogeography 8, 329-345.

Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W.,Baltzer, M., 1998. Complementarity and the use of indicator groups for reserve selection in Uganda. Nature 394, 472-475.

Kerr, J.T., Southwood, T.R.E.,Cihlar, J., 2001. Remotely sensed habitat diversity predicts butterfly species richness and community similarity in Canada. Proceedings of the National Academy of Sciences of the United States of America 98(20), 11365-11370.

Korovin, G., Karpov, E., Isaev, A., Nefedjev, V., Efremov, D., Sedych, V., Sokolov, V., Schmidt, T., Blauberg, K., Eriksson, O.L., Nilsson, S., Raile, G., Sallnaes, O.,Shvidenko, A., 1998. Siberia and Far East Russia's Future Wood Supply: An Analysis. IIASA, Laxenburg.

Laffan, S.W.,Crisp, M.D., 2003. Assessing endemism at multiple spatial scales, with an example from the Australian vascular flora. Journal of Biogeography 30(4), 511-520.

Linder, H.P., 2001. Plant diversity and endemism in sub-Saharan tropical Africa. Journal of Biogeography 28(2), 169-182.

Luck, G.W., Ricketts, T.H., Daily, G.C.,Imhoff, M., 2004. Alleviating spatial conflict between people and biodiversity. Proceedings of the National Academy of Sciences of the United States of America 101(1), 182-186.

MacArthur, R.H.,E.O., W., 1967. The Theory of Island Biogeography. Princeton University Press, Princeton.

Mace, G.M., Balmford, A., Boitani, L., Cowlishaw, G., Dobson, A.P., Faith, D.P., Gaston, K.J., Humphreis, C.J., Vane-Wright, R.I.,Williams, P.H., 2000. It's time to work together and stop duplicating conservation efforts. Nature 405, 393.

Malyshev, L.I., 1975. Kolichestvenny analiz flory: prostranstvennoe raznoobrazie, uroven vidovogo bogatstva i reprezentativnost uchastkov obsledovanija. Botanical Journal 60(11), 1537-1550.

Malyshev, L.I., 1994. Floristic richness of the USSR. In: B.A., Yurtsev (Ed.),, Aktualnye problemy sravnitelnogo izuchenia flor. Nauka, St._Pererburg, pp. 34-87.

Malyshev, L.I., Peschkova, G.A.,Nikiphorova, O.D. (Eds.), 1981-1997. Flora of Siberia. Nauka, Novosibirsk.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Fonseca, G.A.B.,Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Palmén, E.,Newton, C.W., 1969. Atmospheric circulation systems: their structure and physical interpretation. Academic Press, New York.

Petoukhov, V., Ganopolski, A.,Claussen, M., 2003. A Set of Atmosphere Statistical-Dynamical Models: Theoretical Background. PIK, Potsdam.

Pielou, E.C., 1975. Ecological Diversity. Wiley, New York.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C.,Gibbons, D.W., 1993. Rare species, the coincidence of diversity hotspots and conservation strategies. Nature 365, 335-337.

Preston, F.W., 1962. The canonical distribution of commonness and rarity. Ecology 43, 185-215 410-432.

Qian, H., Song, J.-S., Krestov, P., Guo, Q., Wu, Z., Shen, X.,Guo, X., 2003. Large-scale phytogeographical patterns in East Asia in relation to latitudinal and climatic gradients. Journal of Biogeography 30(1), 129-141.

Rahbek, C.,Graves, G., 2001. Multiscale assessment of patterns of avian species richness. Proceedings of the National Academy of Sciences of the United States of America 98(8), 4534-4539.

Ramankutty, N.,Foley, J.A., 1998. Characterizing patterns of global land use: An analysis of global croplands data. Global Biogeochemical Cycles 12, 667-685.

Rosenzweig, M.L., 1995. Species Diversity in Space and Time. Cambridge Univ. Press, Cambridge.

Sala, O.E., Chapin, F.S.I., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Ann Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., LeRoy Poff, N., Sykes, M.T., Walker, B.H., Walker, M.,Wall, D.H., 2000. Global Biodiversity Scenarios for the Year 2100. Science 287, 1770-1774.

Seabloom, E.W., Dobson, A.P.,Stoms, D.M., 2002. Extinction rates under nonrandom patterns of habitat loss. Proceedings of the National Academy of Sciences of the United States of America 99(17), 11229-11234.

Stolbovoi, V.,McCallum, I., 2002. CD-ROM 'Land Resources of Russia'. International Institute for Applied Systems Analysis and the Russian Academy of Science, Laxenburg.

Sugihara, G.,May, R.M., 1990. Applications of fractals in ecology. Trends in Ecology and Evolution 5, 79-86.

Thomas, C.D., Cameron, A., Green, R.E., Bakkenes, M., Beaumont, L.J., Collingham, Y.C., Erasmus, B.F., Ferreira de Siqueira, M., Grainger, A., Hannah, L., Hughes, L., Huntley, B., van Jaarsveld, A.S., Midgley, G.F., Miles, L., Ortega-Huerta, M.A., Peterson, A.T., Phillips, O.L.,Williams, S.E., 2004. Extinction risk from climate change. Nature 427, 145-148.

Tishkov, A., 2002. Biodiversity. In: Stolbovoi, V., McCallum, I. (Eds.), CD-ROM 'Land Resources of Russia'. International Institute for Applied Systems Analysis and the Russian Academy of Science., Laxenburg.

UNEP, 1992. Convention on Biological Diversity.

van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M.W.,Scholtz, C.H., 1998. Biodiversity assessment and conservation strategies. Science 279, 2106-2108.

Venevskaia, I.,Venevsky, S., 2002. Modeling of global vegetation diversity pattern. PIK, Potsdam.

Venevsky, S.,Venevskaia, I., 2003. Energetic and landscape factors of global vegetation diversity. Ecology Letters 6, 1004-1016.

Vetaas, O.R.,Grytnes, J.-A., Distribution of vascular plant species richness and endemic richness along the Himalayan elevation gradient in Nepal. Journal of Biogeography 11(4), 291-301.

Whittaker, R.J., Willis, K.J.,Field, R., 2002. Scale and species richness: towards a general, hierarchical theory of species diversity. Journal of Biogeography 28, 453-470.

Williams, P.H., 2000. Some properties of rarity scores for site-quality assessment. British Journal of Entomology and Natural History 13, 73-86.

Williams, P.H., Lees, D., Araujo, M.B., Humphreis, C.J., Vane-Wright, R.I.,Kitching, I.J., 2002.Biodiversity and Worldmap Ihttp://www.nhm.ac.uk/science/projects/world-map/index.html, The Natural History Museum.

Hierarchical systematic conservation planning for a national level: example of Russia. 2. Setting conservation targets for Russian biodiversity hotspots

Irina Venevskaia, Sergey Venevsky,

Abstract

The aim is to set conservation targets for species found in the three Russian biodiversity hotspots, North Caucasus, South Siberia and Far East in terms of the number of locations where species occur. Species in Red Data book in the five taxons, used in the global biodiversity hotspots analysis (terrestrial mammals, amphibian, reptiles, birds and vascular plants), are selected for setting conservation targets in the hotspots. We propose the percentage targets (baseline and retention), reflecting differential requirements for regional protection, caused by the regional nature of the community structure and specific human threats. We use indicators of species richness/equitability to describe these regional differences in the quantitative conservation targets, required by the systematic conservation planning. We believe, that the majority of countries may apply the hierarchical setting of the conservation targets, because the identification of large-scale biodiversity hotspot requires climate and elevation data only, and national Red Data Books are already elaborated in many countries. We expect, that additional fine-scale data collection and application of process-oriented simulation models may be applied to refine the conservation targets in the case of highly fragmented and diverse regions.

Keywords: National conservation planning, conservation targets, Red Data book species, vascular plants, vertebrate animals, biodiversity hotspots

1. Introduction

The need to provide a clear and practical strategy for biodiversity conservation, which can be transferred from the international to national level, requires as an essential core quantitative algorithms for performing such a transformation.

We showed recently with the example of Russia (Venevsky and Venevskaia, 2004), how national biodiversity hotspots can be identified for a large country from spatial climate, topographic and land use data, by modifying and extending of approach used in the global biodiversity hotspots analysis (Myers et al., 2000). Three national biodiversity hotspots were bordered in Russia, based on the species-energy relationship, which generally reproduces global patterns of vegetation diversity for scales 10000 and 100000 km², and is applicable for different regions across scales from hundred to million km² (Venevsky and Venevskaia, 2003).

Mapping national biodiversity hotspots allows to implement a hierarchical nesting approach for biodiversity conservation in large countries. Here, large-scale regions exposed to anthropogenic pressure, which are nevertheless evolutionary suitable for species persistence and retention, are identified using abiotic factors only at a first stage and then existing and prospective protected areas are analyzed using fine scale species abundance data for different taxons within these hotspots at a second stage.

Here the aim is to set conservation targets for species in the three Russian biodiversity hotspots, North Caucasus, South Siberia and Far East in terms of numbers of localities of actual species distributions. The Red Data book species within the five taxons, also used in the global biodiversity hotspots analysis (Myers et al., 2000) are selected for setting of conservation targets in the hotspots.

2. Formulation of quantitative conservation targets at national level

When national biodiversity hotspots are mapped, a set of biodiversity features to be conserved or protected should be chosen and targets for these features should be formulated. A concept of systematic conservation planning, suggested for global and large-scale regional levels (Margules and Pressey, 2000), requires the use of explicit targets for biodiversity features. The targets provide accountable formulation of broad conservation goals, elaborated by experts and policy makers. Thus, they have inevitably temporal character: after a given time period (e.g. 20-30 years for Cape Floristic Province (Pressey et al., 2003) targets will be reviewed and modified according to new conservation goals.

Two general requirements, agreed by the broad scientific and management public, are applied to design targets in conservation planning (Pressey et al., 2003). Firstly, conservation targets should be quantitative, when possible. Secondly, they should not be constrained by an existing or feasible configuration and size of protected areas, in order to provide a clear picture of the necessary trade-off between land planning and nature conservation. Consideration of both requirements is possible, only when targeted biodiversity features can be mapped or approximated feasibly. Different biodiversity features, such as landscape/land type units, species of some taxa or ecological processes, providing rich diversity, can be in a focus of conservation projects. For instance, the Cape Action Plan for the Environment (Cowling et al., 2003) includes five groups of targeted biodiversity features: 102 broad habitat units; locality records for 364 plant species in the family Proteaceae; locality records for 345 species of reptiles, amphibians and freshwater fish; distributions and densities of 41 species of large and medium-sized animals; and six types of spatial surrogates for ecological and evolutionary processes. Persistence of selected biodiversity features should be provided by scientifically based targets, incorporating both spatial and temporal heterogeneity of biodiversity patterns and anthropogenic threats. There is a visible need to distinguish geographically conservation targets within a fixed group of biodiversity features according to existing patterns of environmental gradients and human impacts. Indeed, the well known conservation targets of 10% or 12% of any geographical areas (McNeely, 1993), including countries, regions or vegetation types, were commonly used for assessing of existing protected areas (Maddock and Benn, 2000), (Sierra et al., 2002), mainly due to the absence of an alternative, widely recognised threshold. However, the 10-12% targets proposed by IUCN are likely to be wholly insufficient for many regions (Rodrigues and Gaston, 2002) and depend on application scale, which is a significant disadvantage for country studies, a domain of which may vary from square kilometres to million of square kilometres. Indeed, protection of 100% of primary vegetation in many regions can be now insufficient for persistence of some bird species due to high fragmentation of habitat and subsequent extinction, lagged behind start of human caused changes by decades and centuries (Brooks et al., 1999). Recent conservation plans (Noss et al., 1999), (Noss et al., 2002), (Pressey et al., 2003) are interpreting suitable composite data to formulate an array of conservation targets according to their environmental and socio-economic status. For example, the Cape Action Plan for the Environment (Pressey et al., 2003) considers environmental gradients by setting an array of baseline targets, while anthropogenic threats are interpreted by the array of retention targets.

Large-scale national biodiversity hotspots identified by broad biogeographical analysis (see for example (Venevsky and Venevskaia, 2004) will differ by climatic, edaphic, topographical conditions and by type and severity of human impacts. Therefore, a methodology for evaluating baseline and retention conservation targets for national biodiversity hotspots should be elaborated, which have to account for structure and type of available biodiversity data.

Only recently has a full description on land use types and ecological processes, in view to preserve biological diversity, with further formulation of targets come into focus of conservation planning on a regional level. Using of species presence/absence and abundance data is rather traditional and based on a considerable amount of botanic and faunistic surveys. Subsets of these surveys, data on rare, threatened and endangered species, listed in Red Data books (RDB), have commonly been retained for setting and implementation of conservation priorities on a national level (for USA, (Abbitt et al., 2000), South Africa (Cowling et al., 1999). Indeed, this biodiversity feature by definition is already at significant risk of loss and, therefore, requires immediate protection measures. Besides, components of the RDB classification explicitly or implicitly indicate level of threatening processes, which can be used for assessment of anthropogenic or climate impact ranges and, thus, for setting of retention targets. A set of rare, threatened and endangered species reflects the upper range of high compositional turnover along climate, edaphic and altitudinal gradients, i.e. reflecting important features of biodiversity in general. The analysis of a complementary reserve network for bird species in South Africa and Lesotho (Bonn et al., 2002), based on data for threatened and endemic species, demonstrated that such a network captures areas, where other species are present, and areas, where the species are abundant. Despite the absence of guaranties in all cases, the study of Bonn et al., (2002) shows that area selection based on threatened and endemic species makes feasible an umbrella for total species diversity on a national level. National Red Data Books provide information about location of best sites for at least some species. This information is important when detailed distribution data are not available for designing a complementary based protection network.

Therefore, we suggest using rare, threatened and endangered species of vascular plants and vertebrate animals that include mammals, reptiles, amphibians and birds as a basis for formulating baseline and retention conservation targets in national biodiversity hotspots. Focusing on these five biodiversity feature groups provides a link between conservation planning for national and global biodiversity hotspots, where much analysis was carried out for the same taxa (Myers et al., 2000), (Brooks et al., 2002).

Two major problems in intra-relations between an international and national conservation planning, based on Red Data books or Red Lists of threatened species, are mismatches between global and national assessments (Rodriguez et al., 2000) (Hilton-Taylor et al., 2000) and data uncertainties on the country level. The criteria, elaborated in 1994 by the IUCN for threatened species (Gärdenfors, 2001) were re-examined and changed between 1997 and 2000 and in a new outlook were never applied on a national level. From all the countries, only Sweden and Finland had applied the old IUCN criteria (IUCN, 1994) for adjusting categories of threat for species at a national level. Despite the general applicability in the Nordic countries of four from the five threat categories identified by IUCN, all the criteria posed several problems due to unrealistic requirements to spatial resolution of basic data or too vague definitions (Gärdenfors, 2001). While conceptual and practical obstacles for direct use of IUCN Red List criteria at a national scale are not yet resolved, an application of country assessment for threatened species can be suggested for conservation planning. The rationale here is that national assessments usually incorporate results of global studies, while the opposite sometimes does not happen (Rodriguez et al., 2000). Another argument for a multiple application of national Red Lists or Red Data books is the existence of threatened transboundary species simultaneously in bordering countries (see example of the USA (Abbitt et al., 2000)), which can be accounted for and featured in the most accurate survey. For example, all the three identified Russian biodiversity hotspots border the countries Georgia, Azerbaidjan, Mongolia and China, where endangered species protection has less comprehensive legislation and assessment than in Russia. Therefore, orientation on the

Russian national Red Data book to formulate conservation goals in the three hotspots will also promote persistence of transboundary species in these countries.

Data uncertainties in species records can be caused by either recorder bias (e.g. in the UK species data collection, (Prendergast et al., 1993) or by design of sampling efforts, based on the preliminary vegetation classification (e.g. in Uganda survey, (Howard et al., 1998). Despite effective methods to remove possible sampling errors (Prendergast et al., 1993), (Howard et al., 1998), national Red Lists and Red Data books may contain errors for species at lower ranges of their abundance and presence. Therefore, the group of threatened species used to formulate conservation goals should be sufficiently large to avoid or the minimize consequences of such shortcomings.

3. Conservation targets for the three Russian biodiversity hotspots

3.1 Data

Russian Red Data books for plants (1984) and animals (1983) contain a significant amount of rather accurate information, which can be used in national conservation planning. The main aim of these RDB is in identifying rare, threatened and endangered species that require urgent protection and conservation measures.

The following information was collected to characterise Russian RDB species: local population size (in number of individuals, or number of metapopulations), geographical range and habitat specificity, level of vitality and threat. A floristic survey for the Red Data books was carried out by specialists of the former Soviet Research Institute of Nature Conservation and Reserves (Beloussova and Denisova, 1981) and a faunistic survey prepared by National Reserves Service of the former USSR (1983).

In this study RDB data for the three identified biodiversity hotspots, North Caucasus, Far East and South Siberia (Venevsky and Venevskaia, 2004), were digitized and put into a spreadsheet database (available as Excel files at www.pik-potsdam.de/~irina/RDBbase) and into geographical information system (available as Arc-Info coverage files at www.pik-potsdam.de/~irina/RDBbase). The database was elaborated for vascular plants, reptiles, amphibian, birds and mammals species. The species were described by their Russian and Latin name, Latin family and genera, category of endemism (endemics or non-endemics on the global level), category of use (hunting animals /medicinal plants or not), transboundary category (names of bordering countries), number of individuals in a hotspot (metapopulations for vascular plants) and area of distribution in a hotspot.

The number of recognized species varies by taxa and hotspot, but not in the same order as area sizes of the hotspots (see Table 1).

Species from the Red Data book	North Caucasus 68 761 km^2	South Siberia 182 875 km^2	Far East 229 691 km^2
Vascular plants	86	20	57
Birds	24	22	37
Mammals	14	8	8
Reptiles	8	0	1
Amphibian	2	0	1

Table 1. Rare, threatened and endangered species in the three hotspots. The areas of the biodiversity hotspots are estimated by (Venevsky and Venevskaia, 2004).

From the total set of 288 RDB species in the hotspots almost sixty percent (171) are the transboundary species, whose distribution areas extend into the neighboring countries of Georgia, Azerbaidjan, Mongolia and China.

Despite of relatively small total area of the three Russian hotspots (they occupy only 3% of the entire Russian territory), these areas are inhabited by 64% of the Russian RDB species belonging to the five chosen taxons.

3.2 Formulating baseline targets for species localities

By definition all the RDB species in the hotspots require urgent conservation and, therefore, should be targeted. Rodrigues and Gaston, (2002) have shown that a reserve network, based on rare taxa, requires slightly greater minimum area than one based on representation of all species. So, representation of all the rare species in conservation targets for the hotspots are also provide representation of other species and the area containing all populations and individuals of rare species seems to be the best solution for conservation planning. However, such a solution, lacking any prioritising scheme, leads to unrealistically large areas. Indeed, environmental gradients between the hotspots result in different composition of ecological communities, characterised by variation in the RDB species richness and evenness. These large-scale differences between hotspots in species representativeness should be evaluated and, then, accounted for in baseline conservation targets. Indeed, preliminary quantitative assessment of species representativeness by hotspots provides data for the relative area requirements in the regions, necessary in the design of baseline conservation targets.

3.2.1 Relative representativeness for the RDB species by hotspot

We suggest to assess the relative representativeness of rare species in hotspots by indicators and models of species richness and evenness (Venevskaia, 1996). Generally, the methodology employed in intra-hotspot comparison of RDB species representativeness follows recommendations for experimental data analyses by Maggurran, (1988) and Southwood, (1978).

Traditionally two main quantitative parameters, number of species (or species richness) and evenness for distribution of species by number of individuals are used to measure overall diversity or representativeness value on a regional level. High evenness for distribution of species by number of individuals is equivalent to a high level of species equitability and, therefore, to overall diversity. An analytical type of function for representativeness (overall diversity) depends on the significance of either species richness or species evenness respectively (Maggurran, 1988), reflected in weighting the indicators of diversity.

The latter can be subdivided into two categories. The first category includes indicators calculated, using total number of species in a selected sample (e.g. Margalef indicator, Schennon indicator, Berger-Parker indicator). The second category comprises indicators obtained from the models of species abundance (e.g. parameters of geometric or logarithmic distributions of species by number of individuals).

We used four indicators of diversity to estimate relative representativness of the RDB species in the Russian hotspots (see Appendix 1 for algorithm of calculations):

- species richness (Margalef indicator)
- species equitability (α parameter of the logarithmic distribution for species by individual numbers)
- combination of species richness and equitability:
    3. Berger-Parker indicator
    4. Shennon indicator or entropy measure.

Direct biological interpretation of the listed indicators could be rather difficult in the case of RDB species, because calculation of indicators is carried out for samples, where the numbers of individuals (or number of populations in the case of plants) may be very small. However, by simultaneous use of the four listed indicators, one can hope to receive quantitatively sound results in comparison of a diversity/represenativeness for large regions (see examples in (Maggurran, 1988). The four indicators for the RDB vascular plants and vertebrate animals were calculated to estimate relative represenativeness of the RDB species in the hotspots. Then discriminating abilities of the four indicators and statistical significance of this discrimination were analysed and averaged values of normalised meaningful indicators were used for formulating baseline conservation targets of national scale conservation planning.

3.2.2 Baseline conservation targets for vascular plants

Despite similar values for the total number of vascular plant species in the three hotspots, i.e. 2501 species in North Caucasus, 2845 and 2641 species in South Siberia and Far East (Venevsky and Venevskaia, 2004), regional diversity of the RDB plant species varies significantly.

The indicators of diversity for rare, threatened and endangered vascular plant species in the Russian biodiversity hotspots are presented in Table 2.

Region	North Caucasus	South Siberia	Far East
Margalef indicator	12.68	3.42	7.96
Berger-Parker indicator	0.049	0.15	0.091
Shennon indicator	4	0.93	3.46
Alpha-parameter of log distribution	37.78	8.82	14.73

Table 2. Species richness and equitability indicators for the RDB vascular plant species in the three Russian biodiversity hotspots.

All indicators of species richness and equitability demonstrate higher values for rare, endangered and threatened vascular plants in North Caucasus, followed by the Far East:

1. The RDB vascular plants richness, estimated by Margalef indicator is significantly higher for North Caucasus, which is most likely correlated to the total number of rare, threatened and endangered vascular plant species in the region.
2. The level of dominance in the three sets of RDB vascular plants, measured by Berger-Parker indicator, is higher for South Siberia, followed by the Far East and North Caucasus hotspots, which have similar indicator value. It means, that equitability of the RDB vascular plant species is high for both North Caucasus and Far East, while the South Siberian set of the RDB vascular plant species is dominated in population number by only few species.
3. We concluded from the entropy measure (Shennon indicator) calculations, that the diversity of the RDB vascular plant species is highest for North Caucasus, followed by the Far East and South Siberia respectively. A statistical *t*-test with the variation of entropy, shows that this composite species richness /equitability indicator differs significantly for the three regions (P>0.99). We did not use this indicator, because its value is equal for North Caucasus and the Far East.
4. The alpha parameter of the log distribution (which is also the composite species richness/equitability indicator) demonstrates discrimination of the three regions similar to the Margalef indicator

The general conclusion of this comparative exercise is that the relative representativness for the RDB vascular plant species in the hotspots can be defined as the average of three indicators, Margalef indicator, the reciprocal to Berger-Parker indicator and the alpha parameter, divided by the values of appropriate indicators for the South Siberian set of species. Indeed, after the division of these three indicators to the values, observed for the

South Siberia hotspot, they became close to each other, inspite of capturing different features (species richness or/and equitability) (see Table 3). The relative representativeness of the RDB vascular plant species in North Caucasus is 3.7 times higher than in South Siberia, followed by the Far East (1.9 times). Most likely the set of RDB vascular plant species in South Siberia region has the lowest representativeness due to less favourable climate conditions in comparison with the other two regions.

Region	North Caucasus	South Siberia	Far East
Margalef indicator	3.71	1.00	2.33
Reciprocal of Berger-Parker indicator	3.06	1.00	1.65
Alpha-parameter of log distribution	4.28	1.00	1.67
Average value (without the Shennon indicator)	3.68	1.00	1.88

Table 3. Relative representativeness for the rare, threatened and endangered vascular plant species in the three hotspots, based on different indicators

The baseline conservation targets for the vascular plant species can be defined in a manner similar to the CAPE study (Pressey et al., 2003) by number of species records in the three set and by weighting the representativeness of a set:

$$B_i = r_i * (M_i - 1) \qquad (1)$$

where B_i is a baseline conservation target in number of populations of vascular plant species in i-th hotspot, r_i is the average of the three indicators (Margalef, reciprocal to Berger-Parker and α parameter) relative representativeness of the hotspot and M_i is the median of number of populations by the RDB species distribution in the hotspot.

For example, M_1 the median for the species by number of population records for vascular plants in the North Caucasus hotspot is equal to 3 records of populations (see Figure 1).

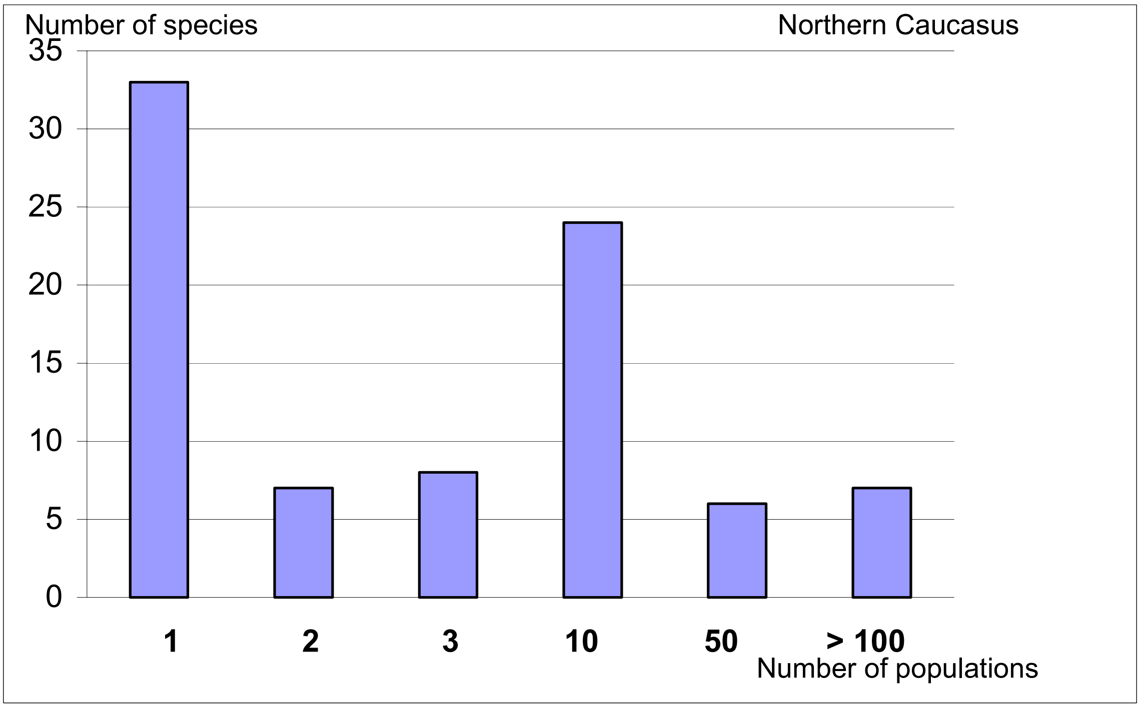


Figure 1. Number of populations for the rare, threatened and endangered vascular plant species in the North Caucasus biodiversity hotspot.

Subtracting 1 from the median M_1 and multiplying the rest by relative representativeness of the hotspot (3.68, see Table 3), we receive the baseline conservation target for North Caucasus (see 1): to represent at least 7 populations of each RDB vascular plant species in a reserve network of the hotspot. If less or equal to seven populations are registered for a vascular plant species from the Red Data book, all their localities should occur in protected areas of the North Caucasus hotspot. The baseline conservation targets, obtained by similar calculation (see equation 1): for South Siberia and the Far East hotspots comprise 3 and 9 populations for each vascular plant RDB species.

The suggested approach in formulating conservation targets aims to protect almost half of the populations of the rare, threatened and endangered vascular plant species as a minimum. It can be easily modified in the frame of the described methodology, if we employ another 'conserve only half' imperative.

3.2.3 Baseline conservation targets for vertebrate animals

Vertebrate animal fauna of Russia is well investigated and numbers over 1300 species. North Caucasus, South Siberia and South of Far East are regions with a high degree of animal species richness and high fauna endemism which can be explained by their historical role as refuges during the Pleistocene and Holocene glaciations (Amirkhanov, 1997).

One hundred and sixty three terrestrial vertebrates of Russia are considered as rare, threatened and endangered species at the national level (Table 4). One hundred and fifteen of them can be found in the identified hotspots, including 82% of amphibian, 75% of reptiles, 72% of mammals and 69% of bird species listed in the Russian Red Data book.

Terrestrial Mammals	Birds	Reptiles	Amphibian
41	107	11	4

Table 4. Rare, threatened and endangered terrestrial vertebrate species of Russia listed in the national Red Data book (1983), (1983).

Representatives of all four classes of Russian rare, threatened and endangered animal species are also listed in the IUCN Red Book, (Baillie and Groombridge, 1996) of which twenty-four land species are found in the three hotspots (see Table 5). This constitutes 89% of the entire set of rare, threatened and endangered species, listed by IUCN for Russia.

Name	Family	Genera	Biodiversity hotspot
	AMPHIBIA		
Pelodytes caucasicus Boulenger	Pelobatidae	Anura	North Caucasus
Onychodactylus fischeri	Hynobiidae	Caudata	Far East
	REPTILES		
Vipera kaznakowi Nikolsky	Viperidae	Serpentes	North Caucasus
	BIRDS		
Paradoxoronis heudei	Panuridae	Passeriformes	Far East
Synthliboramphus wumuzusume Temmnick	Alcidae	Charadriformes	Far East

Anthropoides virgo Linnaeus	Gruidae	Gruiformes	Far East
Grus vipio Pallas	Gruidae	Gruiformes	Far East
Grus japonensis Müller	Gruidae	Gruiformes	Far East
Haliaeetus albicilla Linnaeus	Accipitridae	Falconiformes	Far East, South Siberia, North Caucasus
Aquila heliaca Savigny	Accipitridae	Falconiformes	South Siberia
Mergus squamatus Gould	Anatidae	Anseriforms	Far East
Tadorna crsitata Kuroda	Anatidae	Anseriforms	Far East
Ciconia boyciana Swinhoe	Ciconidae	Ciconiformes	Far East
Nipponia Nippon Temmnick	Threskonithidae	Ciconiformes	Far East
Egretta eulophotes Swinhoe	Ardeidae	Ciconiformes	Far East
	MAMMALS		
Nemorhaedus caudatus Milne-Edwards	Bovidae	Artiodactyla	Far East
Bos bonasus Linnaeus	Bovidae	Artiodactyla	North Caucasus
Panthera pardus orientalis Schlegel	Felidae	Carnivora	Far East
Panthera tigris altaica Temmnick	Felidae	Carnivora	Far East
Panthera pardus tulliana Valenciennes	Felidae	Carnivora	North Caucasus
Uncia uncia Schreber	Felidae	Carnivora	South Siberia
Cuon alpinus Pallas	Canidae	Carnivora	South Siberia

Table 5. Rare, threatened and endangered terrestrial vertebrate species in the biodiversity hotspots, listed in the IUCN Red Book.

The total number of terrestrial vertebrate species from the Red Book of IUCN varies between hotspots: four of these species have their areas in South Siberia, six species are inhabitants of North Caucasus and the fifteen come from the Far East. This difference, however, is not symmetrical between the four different classes. There are eleven IUCN Red Data book bird species in the Far East, making this hotspot exceptionally valuable for international conservation of avian diversity. Distributions of the remaining terrestrial vertebrate species from the IUCN Red Book by classes are very similar between regions.

For consolidation of the international and national conservation efforts we set baseline targets for the 24 vertebrate animal species listed in the IUCN Red Data book to comprise all localities and individuals of these species in the three hotspots.

Baseline targets for the rest of the 91 RDB animal species, found in the hotspots, were formulated taking into account of the overall relative representativeness of these species in the regions by the four classes of interest.

Amphibians and reptiles

Regional data on abundance and geographical distributions of amphibians and reptiles for Russia are either absent or scarce. Lack of data is explained by absence of faunistic inventory for these taxa by the regional state administrative authorities (Ministry of Environmental Protection and Nature Conservation of Russian Federation, 1995).

The total number of reptile species in Russia (75 species) is low due to unfavourable climate conditions over most parts of the territory. It constitutes approximately 1.2% of global diversity in this class of vertebrates (The World Bank, 1995). The richest species diversity of reptiles in Russia is observed in the North Caucasus and in the south of the Far East. About 15% of the reptile species belong to the category of rare and threatened by extinction on a national level, 4% are recorded in the IUCN Red List.

The total number of amphibian species is even smaller (27 species), it constitutes about 0.6% of global diversity in this class (The World Bank, 1995).

The number of rare, threatened and endangered reptile and amphibian species in North Caucasus is 7 reptile/ 2 amphibians, versus 2 and 1 for the Far East and absence in South Siberia. This allows us to hypothesise diversity is highest within these three classes for North Caucasus.

Indeed, here 64% of all Russian RDB reptile species and 50% of all Russian RDB amphibian species are concentrated. There are three North Caucasian endemic species, belonging to these classes, (two of them are listed in the IUCN Red Book), while only one endemic species can be found in the Far East (also listed in the IUCN Red Book).

Data on number of individuals in populations of the amphibian and reptile species in North Caucasus and Far East hotspots are not available. However, they occupy relatively small areas with monospecific landscapes (either semi-arid, or separated by water bodies) in the both regions.

Therefore, we set the baseline conservation target for these species to include all their distribution areas in a nature reserve network for these two hotspots.

Terrestrial mammals

Mammals are the best-known group of vertebrate animals of Russia. The national pool of species accounts for about 7% of their global diversity (The World Bank, 1995). The highest species diversity is a characteristic feature for the North Caucasus, South Siberia, and South Far East (Ministry of Environmental Protection and Nature Conservation of Russian Federation, 1995) (Ministry of Environmental Protection and Nature Resources, 1994).

Maps of geographical distributions of terrestrial mammals (Ministry of Environmental Protection and Nature Conservation of Russian Federation, 1995), were used to calculate the total number of these species in the three hotspots (see Table 6).

Region	North Caucasus	South Siberia	Far East
Number of mammal species	54	58	60

Table 6**.** Total number of mammal species in the three biodiversity hotspots of Russia (Ministry of Environmental Protection and Nature Conservation of Russian Federation, 1995; Ministry of Environmental Protection and Nature Resources, 1994).

Applying similar comparison procedure for the rare, threatened and endangered mammals species as for the RDB plant species (see 3.2.1), we obtain the overall relative representativeness in the hotspots for setting of conservation targets in the hotspots.

Table 7 contains the Margalef indicator, the entropy indicator, the Berger-Parker indicator and the α parameter of the logarithmic distribution for species by individual numbers within the defined taxon.

Region	North Caucasus	South Siberia	Far East
Margalef indicator	0.89	0.8	0.64
Berger-Parker indicator	0.47	0.61	0.52
Shennon indicator	1.3	1.27	1.4
Alpha-parameter of log distribution	6.3	2.9	6.8

Table 7. Species richness and equitability indicators for the rare mammal species in the biodiversity hotspots.

There is no statistically significant difference in the level of species equitability or species richness in the regions, measured by the Margalef indicator, the Berger-Parker indicator and

Shennon indicator. (A statistical test using the *t*-criteria shows that the Shennon indicator for the three regions differs insignificantly, P>0.1).

Only the alpha parameter of the log distribution discriminates the hotspots. However, the samples of mammals are also too small to make this discrimination statistically significant for this indicator.

It means that no priorities can be given to any of hotspots, because of their similar representativeness for the class of mammal species.

We had set conservation targets for the RDB mammal species (except of *Chiroptera* species) in the three hotspots to include of areas with 200 individuals, an estimate of the minimum size needed for a population to avoid inbreeding and other genetic problems (Caughley, 1994). The *Chiroptera* species (bats) occupy a relatively small area in the North Caucasus and Far East, although being rather abundant in their habitats. Consequently, we set targets for the *Chiroptera* species as the total distribution area of these species within the hotspots.

Birds

Birds has been extensively studied in Russia and make up 7.6% of world diversity in this class (The World Bank, 1995).

The geographical distribution of total number of bird species in our regions can be approximately estimated by using maps of richness of bird species (Ministry of Environmental Protection and Nature Conservation of Russian Federation, 1995).

The total numbers of bird species for chosen regions are presented in Table 8.

Region	North Caucasus	South Siberia	Far East
Number of bird species	300	280	380

Table 8**.** Total number of bird species in the three biodiversity hotspots of Russia (Ministry of Environmental Protection and Nature Conservation of Russian Federation, 1995).

The test for relative representativeness for rare, threatened and endangered bird species, has classified the hotspots in a similar order to that for vascular plants: the North Caucasus has the highest value, followed by Far East, with South Siberia in last place (see Table 9).

Region	North Caucasus	South Siberia	Far East
Margalef indicator	1.99	1.22	1.9
Berger-Parker indicator	0.41	0.99	0.49
Shennon indicator	1.6	0.02	1.29
Alpha-parameter of log distribution	2.14	1.25	2.12

Table 9. Species richness and equitability indicators for the rare bird species in the three Russian biodiversity hotspots.

It was found by calculation of species richness and equitability indicators within the class of interest that:

1. Margalef indicator for the North Caucasus hotspot is higher than for the Far East hotspot, despite the larger total number of rare, threatened and endangered bird species in the latter region.
2. The level of dominance for the RDB bird species, measured by the Berger-Parker indicator, is highest for South Siberia and has almost the same magnitude in North Caucasus and the Far East.
3. The diversity of bird species, measured by the Shennon indicator is also highest in the North Caucasus. A statistical test using the *t*-criteria with variations of entropy shows that representativeness of all three regions differs significantly (P>0.9). The small value of Shennon index (as well as largest value of the Berger-Parker indicator) for the South

Siberia hotspot is explained by a overwhelming dominance in the number of individuals of only one species, *Emberizia godlewski Taczanowski,* in the set of the RDB bird species for this hotspot. The extremely large total number of individuals of this species does not allow to use the entropy measure for the calculation of relative representativeness of the RDB bird species in the hotspots.

4. The alpha parameter of log distribution of species by number of individuals distinguishes the hotspots in the defined species group similarly to other indicators.

The comparison exercise allows to formulate baseline targets for the rare, threatened and endangered bird species in the same manner as for the vascular plant species:

$$B^b{}_i = r^b{}_i * (M^b{}_i - 1) \qquad (2)$$

where $B^b{}_i$ is a baseline conservation target in number of individuals in *i*-th hotspot for the RDB bird species, $r^b{}_i$ is averaged by the Margalef indicator, the reciprocal of the Berger-Parker indicator and the α parameter of log distribution relative representativeness of the hotspot in the class of bird species and $M^b{}_i$ is the median of number of individual by species distribution for the RDB bird species in the *i*-th hotspot. Baseline targets for bird species calculated by this formula require presentation of 190 individuals for North Caucasus, 53 individuals for the Far East and 55 for South Siberia to be present in regional nature reserve networks.

3.3 Retention conservation targets for vascular plants and vertebrate animals

Russian Red Data book includes classification of species by category of threat, which allows formulation of retention targets for the rare, threatened and endangered species, used for conservation planning. Categories of human interest (e.g. medicinal plants and hunting or collecting animals), relative abundance (i.e. low number of individuals/populations) and presence/absence of endemism at the global level are identified for each species.

The three categories of threat, listed in the Russian Red Data book for species: 1) to be a rare species, 2) to be an abundant endemic or 3) to be a rare endemic, were used to set the regionally specific retention targets by taxons as the percentages to the baseline conservation targets, defined above (see 3.2). The retention target for each species assumes an increase by a certain percentage in the number of localities, assigned by baseline targets, in order to shield the species from regionally specific natural or human threat of extinction. At first, we set the arbitrary maximum retention targets for the species with direct human use (i.e. medicinal plant or the hunting animals) in relation to the baseline ones to be:

- 150% in case of the rare endemic species,
- 100% in case of the abundant endemics,
- 50% for the rare species.

This formulation generally aims to include in protected areas more than doubled the number of localities for the endemic species, which are permanently under direct threat of elimination because of their medicinal, hunting or collection value.

A threat for the rest of the RDB species is considered by setting maximum retention targets to be two times less than the defined targets for medicinal plants and collected or hunting animals in the three categories, i.e. to range:

- 50-75% in case of the rare endemic species,
- 25-50% in case of the abundant endemics,
- 0-25% for rare species.

The suggested intervals for retention targets facilitate in increase of conservation priorities from the lowest for rare species, which are nationally important, to the highest for rare endemic species, which are important internationally.

Retention targets for a particular RDB species may range, therefore, from 0% to 150% of a baseline target, depending on attributes of species (i.e. human interest, relative abundance and presence/absence of endemism) and geographical location of its distribution area.

Assignment of a retention conservation target for each RDB species from a Russian biodiversity hotspot was carried out in two steps:

Step 1. Category of indirect threat was determined (rare species, abundant endemic or rare endemic species) and a hotspot specific percentage of baseline target in this category was applied to calculate number of required localities for retention of the species.

Step 2. This number of localities was doubled in the case of direct human use of the species.

The regional retention targets for species from each category of threat are set within the percentage interval for this category (i.e. between 50 and 75% in case of the rare endemic species, 25 to 50% in case of the abundant endemics and 0-25% for the rare species). The actual value of a hotspot retention target is defined from the total number of species in this category, inhabiting the hotspot: the hotspots are placed in the percentage interval for the retention target according to their relative species richness in this category.

The maximum retention target for a category is prescribed for a hotspot with the maximum number of species within this category. For example, a retention target of 75% in relation to the baseline is set for the plant rare endemic species of North Caucasus, because the relative number of plant rare endemic species is the highest here in comparison with the two other hotspots. Relative distance in species numbers between hotspots within the defined category is used to place retention targets for the two remaining hotspots into the interval with the length of 25%.

The relative distances between hotspots in the three categories are estimated from the total number of species in a taxon and the RDB data by calculating a regional matrix of rarity (see Appendix 2). All the species in a taxon are sorted by categories of endemism and rarity, defined by the Red Data book, and put into the 2x2 matrix, elements of which are normalised to the total regional species number. The attributes of the rarity matrix are similar to those, proposed by Rabinowitz, (1981): wide or narrow distributional area and small or large size of population.

Elements of the calculated regional matrixes are used for calculating the relative distances in species numbers between the hotspots within categories of threat:

$$\widetilde{P}_j^i = P_j^i * \frac{S^i}{\max_i (S^i)}, \tag{3}$$

where P_j^i is the element of category j in the matrix of rarity, $j=2$ represent rare species, j=3 abundant endemic and j=4 rare endemic species for the i –th hotspot (j =1 describes common species and is not used), S^i is the total number of species in the i –th hotspot, , $0 \le \widetilde{P}_j^i < 1$ is the relative distance in species numbers for the hotspot i in the category j . Division by the maximum total number of species between hotspots in the formula is used to account for the variation in total species pools found in the three hotspots.

The maximum relative distance between the three hotspots within a category j is projected to the upper value for retention target in this category (i.e. 25%, 50% or 75% to the baseline target). The two retention targets for the two remaining hotspots are calculated by placing them into the 25% interval, proportionally to their relative distance in species numbers within the category:

$$R_j^{\,i} = \frac{\widetilde{P}_j^i}{\widetilde{P}_j^{\max}} * 25 + R_j^{\min}, \tag{4}$$

where $R_j^{\,i}$ is the retention target for category j for the i –th region in percentage of the baseline target, $R_j^{\min}$ the lower border of interval set for retention targets in category j (0%

for the category ‘rare species’, 25% for ‘abundant endemics’ and 50% for ‘rare endemics’); $\widetilde{P}_j^{\max} < 1$ is the maximum relative distance between the hotspots in category j.

The matrices of rarity were calculated for the RDB vascular plant, birds and mammal species. Then the ranking procedure, described above, was performed to formulate the retention conservation targets for the species within the three category of threat. Retention targets for the groups of species with an absence of data, or for which areas of distribution are already completely covered by the baseline conservation targets were set to zero. These groups include bats (*Chiroptera* species), amphibian and reptiles.

3.3.1 Retention targets for the RDB vascular plant species

The RDB plant species in all three categories of threat can be met in the Russian biodiversity hotspots. Thus, the described ranking procedure for formulating regional retention conservation targets was applied for the rare non-endemic, the abundant endemics and the rare endemic species categories. The results of the ranking procedure demonstrate exceptional value of the biodiversity hotspot in North Caucasus as the largest number of abundant and rare endemic plant species are located there (see retention conservation targets in percent to the baseline ones in Table 10).

Category of plant species in the matrix of rarity	North Caucasus	South Siberia	Far East
Rare non-endemics (Category 2)	24	7	25
Abundant endemics (Category 3)	50	31	39
Rare endemics (Category 4)	75	54	57

Table 10. Retention conservation targets for the rare, threatened and endangered vascular plant species by a biodiversity hotspot (in percentage of the baseline targets).

The retention targets listed in Table 10 were doubled for medicinal RDB plant species, found in the three hotpots.

3.3.2 Retention targets for the RDB vertebrate animal species

Terrestrial mammals

There are no mammal endemic species, listed in the Russian Red Data Book (1983). On the other hand, the relative distances in species numbers (see equation 3) are almost equal between the three hotspots (0.1 for North Caucasus, 0.13 for Far East and South Siberia) for the category 2 (rare species), when the *Chiroptera* species are not considered. Thus, we set the same retention targets for the mammal species from our subset in the three Russian biodiversity hotspots. These are equal to 50 individuals (25% of the baseline target), when the species are not hunting, and to 100 individuals (50% of the baseline target) in the opposite case.

Birds

Only the Far East biodiversity hotspot includes the six abundant endemic bird species, represented in the national Red Data book. Retention target in this category is set to 27 individuals (50% of the baseline target for the bird species at Far East) in a regional reserve network. This retention target was doubled (53 individuals) for *Grus monacha* and *Mergus squamatus* as the hunting endemic bird species.

The ranking procedure (see equations 3,4) in the category of the non-endemic RDB bird species, resulted in formulating a maximum retention conservation target of 25% to the baseline target for the Far East hotspot (13 individuals), 21% (39 individuals) for North Caucasus and 19% (10 individuals) as the retention conservation target for South Siberia. The retention targets were doubled, if the species were directly endangered by humans (almost 50% of the RDB bird species in the three hotspots are hunted).

3.4 Final targets

3.4.1 Conservation targets for the RDB vascular plant species

Final targets for the rare, threatened and endangered vascular plant species vary between 25% and 100% from their recorded localities. Almost all of the RDB plant species from the biodiversity hotspot in North Caucasus (76 species from the 86) are required 100% of their recent distribution area in a targeted reserve network (see Figure 2). Reserve networks in the hotspots in South Siberia and the Far East should include the entire areas of distribution for over a half of the RDB plant species.

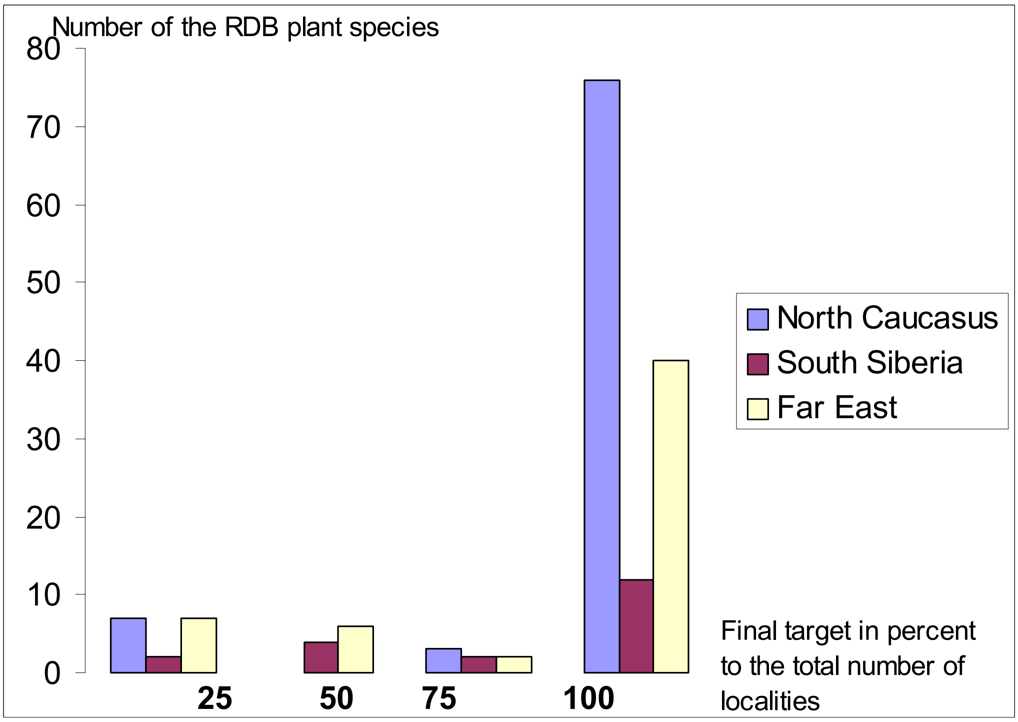


Figure 2. Distribution of targets as percentages of total records by the RDB plant species in the Russian biodiversity hotspots.

3.4.2 Conservation targets for the RDB vertebrate animal species

The conservation targets for the RDB animal species for the three Russian biodiversity hotspots range from 100% of the total areas in a reserve network to just few recorded locations. (see Figure 3). Unlike the conservation targets for the RDB vascular plant species (see Figure 2), the final targets for the animal species have similar distributions across all the three hotspots, revealing no leading role for any region in required protected areas. Indeed, between twenty to thirty species in the three hotspots require conservation of their entire areas of distribution, and these species comprise more than half the RDB vertebrate animal species found in the regions.

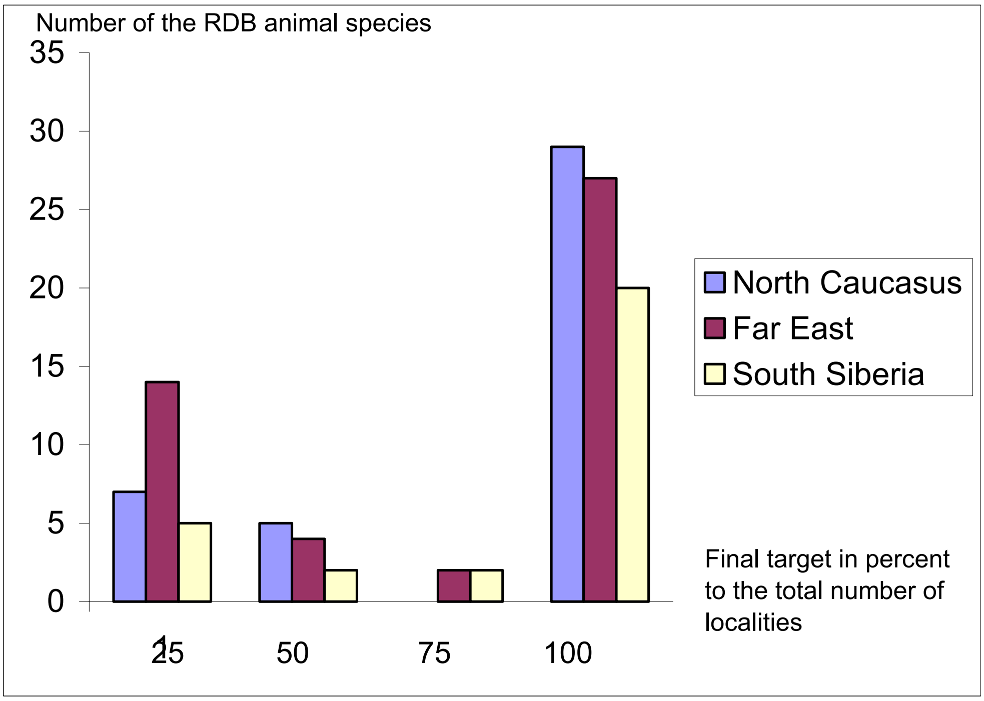


Figure 3. Distribution of targets as percentages of total areas by the RDB vertebrate animal species in the Russian biodiversity hotspots.

4.Discussion

Formulation of explicit national conservation targets involves interpretation of verbal goals (like " best representation and maintenance of the national biodiversity with the smallest required area") through the filter of available data on the biological diversity and threats to it. We suggested hierarchical approach, when large-scale national biodiversity hotspots are identified in the first stage (Venevsky and Venevskaia, 2004), using the simulated data for species number of vascular plants only, and quantitative conservation targets for the five groups of the Red Data Book species (terrestrial mammals, amphibian, reptiles, birds and vascular plants) are defined in the second stage. Such an application of the hierarchical approach is required by the systematic conservation planning (Cowling et al., 1999) to underpin hierarchical nature of biodiversity, features of which will be described with visible flaws in any single-scale conservation assessment (Fairbanks and Benn, 2000). Individual taxonomic groups (the vascular plant species of the national biodiversity hotspots analysis (Venevsky and Venevskaia, 2004), can represent at their best only approximate surrogates for the distribution of other taxons (Howard et al., 1998). This is why we applied the analysis for the RDB species from the five taxonomic groups, which are focus of the international conservation planning (Myers et al., 2000). A combination of the national pattern for the vascular plant species (Venevsky and Venevskaia, 2004) and the regional patterns for the RDB vascular plant species and the RDB vertebrate animal species allows to represent at the same time a contribution of commonness and rarity of species, both necessary "to quantify" the overall national biodiversity (Lennon et al., 2004) and, thus, to quantify the conservation targets. We considered that the baseline and the retention targets, should be geographically distributed to reflect differential requirements for regional protection, caused by the regional structure of the communities and regionally specific human threats. We suggest to use indicators of the species richness/equitability to describe these regional differences in the quantitative conservation targets, required by the systematic conservation planning (Pressey et al., 2003). To what extent this approach in setting conservation targets could be adapted for countries other then Russia? We believe, that the majority of countries can apply it, because the identification of large-scale biodiversity hotspots require climate and elevation data only, and national Red Data Books are already elaborated in many countries (Gärdenfors, 2001). We, however, are aware that more sophisticated methods for setting conservation targets are appropriate in the regions where biodiversity and abiotic data are accurate, has fine resolution and complete coverage (e.g. setting the conservation targets for the Cape Floristic Province (Pressey et al., 2003). This is especially important for highly fragmented and diverse regions (like North Caucasus from the set of Russian national biodiversity hotspots), where the difference in projected conservation efficiency values may achieve two times, when using fine-scale data based protected areas instead of broad-scale data based reserve networks (see comparison of broad-scale and fine-scale conservation analysis for Agulhas Plain, South Africa (Rouget, 2003). Additional fine-scale data collection and application of simulation models (e.g. population viability analysis for endangered species, Ralls et al., (2002)) may become necessary to feature ecological processes, providing biodiversity, in such highly fragmented regions.

Acknowledgments

We acknowledge Franz Badeck and Dieter Wallschlaeger for providing an opportunity to conduct this study and Stephen Sitch for editing of the paper.

APPENDIX 1. Calculation of diversity indicators for large regions

Species richness

Margalef indicator (Clifford and Stephenson, 1975) was used to evaluate the RDB species richness:

$$M = \frac{(S-1)}{\ln N}, \tag{1}$$

where N is the total number of individuals of the RDB species, S is the total number of the RDB species in a region.

The value of N is the highest source of uncertainty in the Margalef indicator. We propose to use the number of populations for rare plants, defined in the Russian Red Book (1984), instead of number of individuals, which is practically impossible to obtain. For vertebrates one can use expert estimates of individuals numbers, presented in the Russian Red Book (1983), as well as in the local Red Lists for North Caucasus (1990) and for Far East (Ler, 1989).

Species equitability

Evenness indicators are used to investigate whether number of individuals for different species varies considerably, or practically does not change across species. The analysis of evenness is carried out by analysis of observed number of individuals (number of populations for vascular plants) by species distributions. The following statistical distributions are used to approximate these distributions: a geometrical distribution (Whittaker, 1972), a logarithmic distribution and stochastic niche boundaries distribution (MacArthur and Wilson, 1967).

A logarithmic distribution, used in our study, was proposed by Fisher et al., (1943) as the first attempt to describe relations between species and individuals in an ecological community. The logarithmic distribution shows a stochastic process, in which species invasion in a definite site occurs with non-equal intervals of time (May, 1975). The number of species with a population of *n* individuals can be approximated by logarithmic distribution as:

$$\alpha * x, \frac{\alpha * x^2}{2}, \frac{\alpha * x^3}{3}, ... \frac{\alpha * x^n}{n} \quad . \tag{2}$$

The coefficients α and x can be calculated by solution of the system with two non-linear algebraic equations:

$$\frac{S}{N} = (1-x)/x * (-\ln(1-x)) \tag{3}$$

$$\alpha = \frac{N * (1-x)}{x}, \tag{4}$$

where N is the total number of individuals of the RDB species, S is the total number of the RDB species in a region. The coefficient α has a good discriminating ability for comparison purposes and a low sensitivity to the sample size (Taylor, 1978), which is very important for an assessment of RDB species diversity.

The statistical significance for approximation by logarithmic function (2) of the observed number of individuals by the RDB species distributions was determined by χ^2 and *t*-statistics tests.

Species richness/equitability

There are several non-parametric indicators (Southwood, 1978), which do not require assumptions on the type of statistical distributions. They originate from information theory (information measures) or from simple geometrical assumptions about distances between species (measures of dominance).

We use two such indicators in our comparative exercise:

- The Shennon-Winer indicator (informational entropy):

$$H = -\Sigma \frac{n_i}{N} * \ln(\frac{n_i}{N}), \tag{5}$$

where n_i is the number of individuals of species i.

It is assumed from information theory, that the higher the entropy, the more diverse is the set of the RDB species in a region.

The uncertainty range for the Shennon-Winer indicator for a set of the RDB species in a region was estimated by calculating the variation of the informational entropy (Hutchenson, 1970):

$$D = \frac{\Sigma \frac{n_i}{N} * (\ln(\frac{n_i}{N}))^2 - (\Sigma \frac{n_i}{N} * (\ln(\frac{n_i}{N}))^2}{N} + \frac{S-1}{2N^2} \tag{6}$$

- The easiest way for calculations Berger-Parker indicator (Berger and Parker, 1970):

$$d = N / N_{\max}, \tag{7}$$

where $N_{\max}$ is the number of individuals of the most frequent species in the set of the RDB species. The higher the reciprocal of the indicator, the more even are the species in terms of number of individuals.

APPENDIX 2. Regional matrix of rarity

All the species, belonging to one of the three taxons (terrestrial mammals, birds or vascular plants) in a biodiversity hotspot, were divided into the following categories:

- Category 1. Species with wide area distribution and with large local populations (measured either in number of individuals or number of metapopulations). These are common species (not the RDB species) found in a region.
- Category 2. Species with narrow area distribution and large local populations. These are relatively abundant endemics, which can be damaged easily by diminishing habitat areas due to human or natural reasons.
- Category 3. Species with wide area distribution and a small local populations. These are regionally endangered and threatened species.
- Category 4. Species with narrow area distribution and small local populations (or a small number of local populations in the case of plants). These are rare endemics, which are sensitive with respect to both their range and fluctuations in numbers.

We produced a matrix S with the species numbers according to the listed categories:

	Wide area	Narrow area
High size of populations	S_1	S_2
Small size of populations	S_3	S_4

where S_i is the number of species belonging to the defined category *I.*
By dividing the matrix elements by $S_{total} = S_1 + S_2 + S_3 + S_4$, we receive a normalized or a probability matrix of rarity *P*.

$$P = \begin{bmatrix} p_1 & p_2 \\ p_3 & p_4 \end{bmatrix} = \begin{bmatrix} \frac{S_1}{s} & \frac{S_3}{s} \\ \frac{S_2}{s} & \frac{S_4}{s} \end{bmatrix} \tag{1}$$

The elements of the probability matrix of rarity could be interpreted in the following manner:
p_1- is the probability of event, that a species taken from our full set of species is *not a rare, endangered or threatened* one.
1- p_1-is the probability, that a species taken from our full set of species is *a rare, endangered or threatened* one.
$p_2 + p_4$ is the probability, that a species taken from our full set of species has a narrow area distribution (endemic of the region), or is a *rare and threatened* one.
$p_3 + p_4$ is the probability, that a species taken from our full set of species has a small size of population, or is a *rare and endangered* one.
p_4 is the probability, that a species taken from our full set of species has a narrow area distribution (endemic of the region) and a small population (rare endemic of the region), or is a *rare* one.
The value of p_4 is an important quantitative characteristic of the region, as species from this category have the highest risk of extinction.

References

Abbitt, R.J.F., Scott, J.M.,Wilcove, D.S., 2000. The geography of vulnerability: incorporating species geography and human development patterns into conservation planning. Biological Conservation 96(2), 169-175.

Amirkhanov, A.M. (Ed.), 1997. Biodiversity conservation in Russia, The first national report of Russian Federation., Moscow.

Baillie, J.,Groombridge, B. (Eds.), 1996. 1996 IUCN Red List of Threatened Animals. IUCN, Gland.

Beloussova, L.,Denisova, L., 1981. The USSR Red Data Book and its compilation. In: Synge, H. (Ed.),, The biological aspects of rare plant conservation. John Wiley & Sons Ltd., Chichester, pp. 93-99.

Berger, W.H.,Parker, F.L., 1970. Diversity of planktonic Foraminifera in deep sea sediments. Science 168, 1345-1347.

Bonn, A., Rodrigues, A.S.L.,Gaston, K.J., 2002. Threatened and endemic species: are they good indicators of patterns of biodiversity on a national scale? Ecology Letters 5, 733-741.

Brooks, T.M., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Rylands, A.B., Konstant, W.R., Flick, P., Pilgrim, J., Oldfield, S., Magin, G.,Hilton-Taylor, C., 2002. Habitat loss and extinction in the hotspots of biodiversity. Conservation Biology 16(4), 909-923.

Brooks, T.M., Pimm, S.L.,Oyugi, J.O., 1999. Time lag between deforestation and bird extinction in tropical forest fragments. Conservation Biology 13, 1140–1150.

Caughley, G., 1994. Directions in conservation biology. Journal of Animal Ecology 63(2), 215-244.

Clifford, H.T.,Stephenson, W., 1975. An Introduction to Numerical Classification. Academic Press, London.

Cowling, R.M., Pressey, R.L., Lombard, A.T., Desmet, P.G.,Ellis, A.G., 1999. From representation to persistence: requirements for a sustainable system of conservation areas in the species-rich mediterranean-climate desert of southern Africa. Diversity and Distributions 5, 51-57.

Cowling, R.M., Pressey, R.L., Rouget, M.,Lombard, A.T., 2003. A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa. Biological Conservation 112, 191-216.

Fairbanks, D.H.,Benn, G.A., 2000. Identifying regional landscapes for conservation planning: a case study from KwaZulu-Natal, South Africa. Landscape and Urban Planning 50(4), 237-257.

Fisher, R.A., Corbet, A.S.,Williams, C.B., 1943. The relation between the number of species and the number of individuals in a random simple of an animal population. Journal of Animal Ecology 12, 42-58.

Gärdenfors, U., 2001. Classifying threatened species at national versus global levels. Trends in Ecology & Evolution 16(9), 511-516.

Hilton-Taylor, C., Mace, G.M., Capper, D.R., Collar, N.J., Stuart, S.N., Bibby, C.J., Pollock, C.,Thomsen, J.B., 2000. Assessment mismatches must be sorted out: they leave species at risk. Nature 404, 541.

Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W.,Baltzer, M., 1998. Complementarity and the use of indicator groups for reserve selection in Uganda. Nature 394, 472-475.

Hutchenson, K., 1970. A test for comparing diversites based on the Shannon formula. Journal of Theoretical Biology 29, 151-154.

IUCN, 1994. Red List Categories and Criteria. IUCN Species Survival Commission.

Lennon, J.J., Koleff, P., Greenwood, J.J.,Gaston, K.J., 2004. Contribution of rarity and commonness to patterns of species richness. Ecology Letters 7(2), 81-87.

Ler, P.A. (Ed.), 1989. Rare vertebrates of the Soviet Far East and their protection. Nauka, Leningrad.

MacArthur, R.H.,Wilson, E.O., 1967. The Theory of Island Biogeography. Princeton University Press, Princeton.

Maddock, A.,Benn, G.A., 2000. Identification of conservation-worthy areas in northern Zululand, South Africa. Conservation Biology 14(1), 155-166.

Maggurran, A., 1988. Ecological Diversity and its Measurement. Croom Helm, London.

Margules, C.R.,Pressey, R.L., 2000. Systematic conservation planning. Nature 405, 243-253.

May, R.M., 1975. Patterns of species abundance and diversity. In: Cody, M.L., Diamond, J.M. (Eds.), Ecology and Evolution of Communities. Harvard University Press, Cambridge, pp. 81-120.

McNeely, J.A., The Biodiversity Convention since UNCED. 1993 Econet Biodiversity conference, Gland, Switzerland.

Ministry of Environmental Protection and Nature Conservation of Russian Federation, 1995. The State report of environmental status of Russia in 1994, Moscow.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Fonseca, G.A.B.,Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Noss, R.F., Carroll, C., Vance-Borland, K.,Wuerthner, G., 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem. Conservation Biology 16(4), 895-908.

Noss, R.F., Strittholt, J.R., Vance-Borland, K., Carroll, C.,Frost, P., 1999. A conservation plan for the Klamat-Siskiyou ecoregion. Natural Areas Journal 19(4), 392-411.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C.,Gibbons, D.W., 1993. Rare species, the coincidence of diversity hotspots and conservation strategies. Nature 365, 335-337.

Pressey, R.L., Cowling, R.M.,Rouget, M., 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. Biological Conservation 112, 99-127.

Rabinowitz, D., 1981. Seven forms of rarity. In: Synge, H. (Ed.),, The biological Aspects of Rare Plant Conservation. John Wiley, Chichester, pp. 205-217.

Ralls, K., Beissinger, S.R.,Cochrane, J.F., 2002. Guidelines for using population viability analysis in endangered-species management. In: Beissinger, S.R., McCullough, D.R. (Eds.), Population Viability Analysis. University of Chicago Press, Chicago, IL, pp. 521-550.

Red Data Book for Northern Caucasus, 1990. Rostov University Press, Rostov-on-Don.

Red Data Book of Russian Federation / Plants, 1984. Rosselhozizdat, Moscow.

Red Data Book of the Russian Federation / Animals, 1983. Rosselhozizdat, Moscow.

Rodrigues, A.S.L.,Gaston, K.J., 2002. Optimisation in reserve selection procedures - why not? Biological Conservation 107(1), 123-129.

Rodriguez, J.P., Ashenfelter, G., Rojas-Suarez, F., Fernandez, J.J.G.,Suarez, L., 2000. Local date are vital to worldwide conservation. Nature 403, 241.

Rouget, M., 2003. Measuring conservation value at fine and broad scales: implications for a diverse and fragmented region, the Agulhas Plain. Biological Conservation 112(1-2), 217-232.

Sierra, R., Campos, F.,Chamberlin, J., 2002. Assessing biodiversity conservation priorities: ecosystem risk and representativeness in continental Ecuador. Landscape and Urban Planning 59(2), 95-110.

Southwood, T.R., 1978. Ecological Methods. Chapman & Hall, London.

Taylor, L.R., 1978. Bates, Williams, Hutchinson - variety of diversities. In: Mound, L.A., Waloff, N. (Eds.), Diversity of Insect Faunas. Blackwell, Oxford, pp. 1-18.

The World Bank, 1995. Priorities for investment: action plan for protected area system of Russia., Washington.

Venevskaia, I.V., 1996. State of species diversity at the Northern Caucasus in comparison with other regions of Russia and some steps in biodiversity management in the region. Master of Science Thesis, CEU, Budapest.

Venevsky, S.,Venevskaia, I., 2003. Energetic and landscape factors of global vegetation diversity. Ecology Letters 6, 1004-1016.

Venevsky, S.,Venevskaia, I., 2004. Hierarchical systematic conservation planning for a national level: example of Russia. 1. Identifying national biodiversity hotspots using abiotic factors.

Whittaker, R.H., 1972. Evolution and measurement of species diversity. Taxon 21, 213-251.

Hierarchical systematic conservation planning for a national level: example of Russia. 3. Large scale conservation plan for Russian hotspots

Irina Venevskaia, Sergey Venevsky

Abstract

We suggest a large-scale national conservation plan for the three biodiversity hotspots North Caucasus, South Siberia and the Far East. The aim of our conservation plan is the 100% achievement of identified conservation targets for the Red Data book species in one of four groups (vascular plants, amphibian/reptiles, mammals, birds) in every hotspot, while maximising the percentage of target achievement for the other groups with a minimum area requirement. The existing statutory reserves were analysed as regards their environmental bias and efficiency of species representation in view of the conservation targets. It was shown that the protected areas are environmentally biased for the North Caucasus and South Siberia and only a third (at maximum) of conservation targets set for the Russian biodiversity hotspots is achieved within the existing reserve networks. The new large-scale conservation areas within the hotspots are designed by formal optimisation procedure. Implementation issues for the new large-scale conservation plan for Russia, including scheduling, forms of protection, costs and further refining of the plan are discussed.

Keywords: National conservation plan, protected areas, achievement of conservation targets, Russia, biodiversity hotspots

1. Introduction

Management of biodiversity on a national level often fails to coincide with international efforts and even on an international level a limited consensus has so far been achieved on biodiversity conservation priorities (Mace et al., 2000).

Many costly programmes with similar goals have been carried out recently by variety of international organizations, such as Conservation International (Myers et al., 2000), WWF and IUCN (WWF and IUCN, 1994-1997), and the World Resources Institute (Ayensu et al., 1999). The duplication of investigation and management efforts on the international level, leading to competition instead of complementary priority setting, can be explained to a certain extent by conceptual difficulties in biodiversity conservation (see discussion in Pimm and Lawton, (1998)). Indeed, different conservation targets may be at issue, including total species richness for certain taxa, endemic species for a taxa, some particular species of public interest, characteristic landscapes or even environmental functions of ecosystems and genetic pools of populations. The traditionally most-developed branch of biodiversity conservation, maintaining and preserving the variety of species, demands considerable investigation and policy efforts to develop strategies, based on scientific principles. Precise assessments of species richness, the area of habitat remaining, the level of threat to biological diversity, and the risk of extinction in conditions of rapid societal change are crucial questions for quantifying biodiversity and consequently for defining conservation policy.

Conservation decisions are strongly influenced by the geographical scale of assessment. Moreover, conservation priorities identified for one taxa may fail for another taxa. Preserving present-day biodiversity patterns is not a guarantee that ecological and evolutionary processes can be stabilised in the future. The urgency to produce a clear and practical strategy for biodiversity conservation, which is necessary to guide decision-makers on international and national levels, is widely recognised (Mace et al., 2000; Fonseca et al., 2000) and is currently being discussed among the scientific community.

Considerable progress has been achieved recently in developing systematic conservation planning principles (Margules and Pressey, (2000). The geographical objects of systematic conservation planning should be existing or prospective sets of protected areas, which are supposed to represent or sample biodiversity, ideally on all levels of biological organisation, and to promote long-term persistence of elements of biodiversity (ecosystems, species or populations). However, it is now recognised that existing sets of reserves (strictly protected areas of IUCN (1994) categories I –IV) are inadequate in representing biodiversity both in the least developed regions, e.g. New Caledonia (Jaffre et al., 1998), and in regions with long conservation traditions, like Australia (Pressey et al., 2000). The insufficient performance of existing reserve networks is mirrored by the history of national conservation systems, established along the lines of different ideological and economic considerations. Such considerations have led to protected areas often tending to be concentrate in remote regions or on land with low economic value (Scott et al., 2001).

Overcoming shortcomings in the existing reserve networks of a region needs a systematic iterative approach (Margules and Pressey, 2000), consisting of four analysis and two management stages, respectively. The analysis stages include measuring and mapping biodiversity, setting of quantitative conservation targets, reviewing existing protected areas on the basis of these targets, and selection of additional conservation areas.

An application of the hierarchical approach is required for systematic conservation planning (Cowling et al., 1999) to underpin the hierarchical nature of biodiversity.

For instance, the mapping of biodiversity hotspots allows implementation of a hierarchical nesting approach for biodiversity conservation in large countries. Large-scale regions suitable for species persistence and retention, but which are exposed to anthropogenic pressure (hotspots), are identified at a first stage, and at the second stage existing and prospective protected areas are analysed using fine scale species abundance data for different taxa within these hotspots.

In our previous study we applied this approach for Russia, where during the first stage large-scale national biodiversity hotspots were identified (Venevsky and Venevskaia, 2004) using simulated data for species numbers of vascular plants, and quantitative conservation targets for the five groups of the Red Data Book species (terrestrial mammals, amphibian, reptiles, birds and vascular plants) were defined at the next stage (Venevskaia and Venevsky , 2004).

Three Russian biodiversity hotspots, North Caucasus, South Siberia and the Far East, comprising approximately 3% of the entire area of the country, were mapped. The resulting hotspots cover national-scale environmental gradients in Russia and have also been identified by Russian experts, but without specifying actual areas.

The Russian Red Data Book (RDB) species in the five taxa used in the global biodiversity hotspots analysis were selected in order to set spatially distributed conservation targets in the hotspots. We used the indicators of the species richness/equitability to describe the regional differences in the quantitative conservation targets and elaborated the set of required number of localities/areas for persistence and retention of each of the RDB species.

A next key stage of the systematic conservation planning is the review of existing protected areas and elaboration of a new spatial pattern for reserve networks. This stage, carried out in this study, involves estimation of the fulfillment of quantitative conservation targets by the existing reserve networks in Russian biodiversity hotspots, and the planning of new conservation areas based on this analysis.

Russia has a more than century-long tradition in biodiversity conservation, the first protected area in Ascania Nova (now Ukraine) being organised in 1882 (Sokolov et al., 1988).

However, in Russia as in many other countries, conservation areas have often been established within reasonably large geographically homogeneous units, i.e. without consideration of quantitative conservation targets for persistence and retention of national or regional biodiversity. A consequence of such „ad hoc reservation“ (Pressey, 1994) in many cases is a systematic bias of candidate conservation areas away from landscapes with high opportunity costs for agriculture, commercial forestry or urban development, towards remote and scenic regions (Pressey et al., 1996).

A significant effort has been made in developing and refining techniques to identify a representative system of conservation areas in the former Soviet Union (Sokolov et al., 1988). However, serious changes have occurred recently in Russia, forcing an adjustment regarding the national reserve networks to approaches suggested by the conservation planning system. Russia has undergone considerable socio-economic changes in the last 15 years, which have resulted in:

- substitution of the large set of Soviet conservation areas by a smaller one, situated within the territory of the Russian Federation
- increased anthropogenic pressure due to economical and political migration to the regions with high biodiversity (e.g. the Far East and North Caucasus)
- decrease in the competitive abilities of protected areas in comparison to other commercial types of land use, due to lack of state and private funding.

We suggest elaboration of a new national conservation plan for Russia, which will be based on principles of systematic conservation planning (Margules and Pressey, 2000). In this plan the existing reserve networks in the three Russian biodiversity hotspots are analysed by an assessment of the extent to which the regionally specific baseline and retention conservation targets (Venevskaia and Venevsky , 2004) are already achieved, the environmental bias of the existing protected areas is quantified, and the areas for new optimal reserve schemes are proposed for the hotspots. These new conservation areas are elaborated by a formal optimisation procedure, which maximises the fulfilment of the targets within the minimum area required, i.e. the criteria widely used in systematic conservation planning (Araujo, 1999; R.M. Cowling et al., 2003).

2. Review of existing protected areas in the national biodiversity hotspots

2.1 National reserve network of Russia

The Russian national reserve network (NRN) includes ten major types of protected areas, which are managed by different federal and local state authorities. Statutory conservation areas, supported by strong legislative and management infrastructure, are federal natural *zapovedniks*, national parks, and natural landmarks, which are classified by IUCN as protected areas of category Ia and II (www.biodat.ru).

Of the ten types of protected areas in Russia, probably the only type that is fully developed and oriented to the conservation of rare, threatened and endangered species *in situ* is the *zapovedniks*. These protected areas are managed by the authorities of the Department of Protected Areas at the Ministry of Natural Resources of the Russian Federation as a rule, or by the Russian Academy of Sciences. At end of 2000 the number of state *zapovedniks* had reached the number of 100 (Tishkov, 2002), with a total area constituting 1.55% of the whole territory of Russia.

All kinds of economic activity are prohibited in these protected areas and only research or monitoring activities can take place. Thirty of the Russian *zapovedniks* are recognised by UNESCO as Biosphere Reserves from the EuroMAB network (www.unesco.org/mab/)

A state system of Russia Federation national parks has been established since 1983. There were 35 nationals parks with the total area of 69,27 thousand km² (0.4% of the entire territory) by 2001 in Russia (Tishkov, 2002). National parks are designated not only for nature protection purposes, but also to promote scientific, educational and cultural programmes. Controlled tourism activities are allowed in this category of protected areas. Besides this, the allocation of customary extensive nature-use zones is acceptable in the national parks, linked with areas designated to aboriginal communities. Consequently, national parks play a lesser role in species conservation than *zapovedniks* and are defined as protected areas of IUCN category II (www.biodat.ru).

There were thirty federal-level natural landmarks and more than eight thousand regional and local-level-landmarks by year 2000 in Russia (Tishkov, 2002). Nature complexes and objects of natural or artificial origin, which are unique in their ecological, scientific and cultural features, constitute this category of protected areas. Natural landmarks as a rule do not have significantly large areas and could be considered as nucleuses for potential protected areas, important for the conservation of endemic species.

There are also 66 federal-level *zakazniks*, with the total area about 17 000 thousand km². Most of them are managed by the Department for Protection and Rational Use of Hunting Resources under the Russian Ministry of Agriculture (Ministry of Environmental Protection and Nature Conservation of the Russian Federation, 1995). The areas of federal natural *zakazniks* have a particular value for the conservation or recovery of natural complexes and their components and for a sustainable ecological balance. They can also comprise a basis for future statutory reserves. It should be noted that, as a rule, they are organised for the regulation of the numbers of commercial game species, but not for the protection of rare, threatened and endangered species. According to the IUCN classification *zakazniks* belong to protected areas with category III and IV.

The total share of NRN in the land resources of Russia currently amounts to 2.1% of the area whole country (Tishkov, 2002). This value is similar to the share of NRN in the republics of the former USSR, because the absolute majority of protected areas were established during the Soviet time. For instance, the former strictly and non-strictly protected areas in the neighbouring Caucasian republic of Georgia covered 3.2% of the country's territory (www.biofor.com/documents/ Georgia-Biodiversity-Report.pdf). However, the NRN of Georgia is currently under reorganisation in anticipation of a broad-based privatisation of state lands. The old system of *zapovedniks* here was found inadequate and inflexible by the Georgian government in 1990 and a new NRN system is under construction, with a possible general lowering of protection status and decentralisation of reserve management.

The management and legislations scheme of the federal statutory reserves in Russia remains untouched, despite a visible lack of funding. Moreover, Russia plans to increase the NRN to more than 3% of the entire territory, according to the multilateral environmental agreements signed by the USSR in the period 1950-1990 and later ratified by the Russian parliament (Nikitina, 1995).

The non-statutory conservation areas in Russia are represented by 1600 natural reserves and hunting grounds (*zakazniks*) of local and regional jurisdiction.

Despite the large number of these reserves, they do not play a significant role in conservation in Russia due to a lack of personnel and funding, and because nature protection regulation is not fully implemented at the municipal level. We therefore consider in our study only protected areas of the IUCN categories I and II, i.e. those coming under federal jurisdiction.

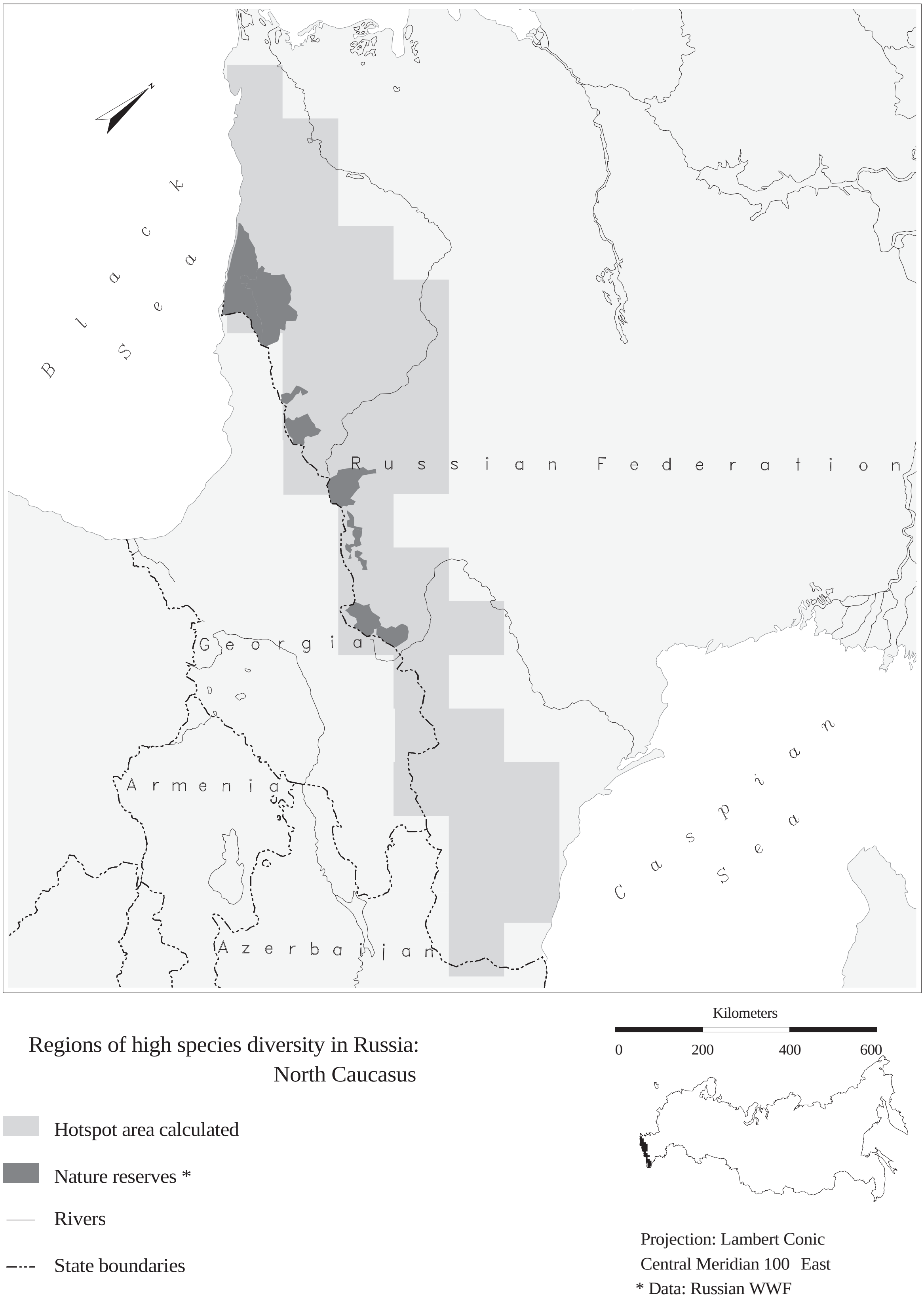
2.2 Protected areas in hotspots

The three Russian biodiversity hotspots, North Caucasus, South Siberia and the Far East, identified in the first stage of our conservation planning study (Venevsky and Venevskaia, 2004), are covered by statutory nature reserves to varying extents (see Figure 1 and Table 1).

Statutory reserves in the hotspots	Area (km^2)
North Caucasus	68,761
Alania National Park	550
Kabardino-Balkarsky zapovednik	825.07
Kavkazsky zapovednik	2,803.35
Prielbrus'e National Park	1,004.00
Severo-Osetinsky zapovednik	295
Sochinsky National Park	1,937.37
Teberdinsky zapovednik	849.96
	Total for hotspot 8,264.75
South Siberia	182,875
Altaisky zapovednik	8,812.38
Azas zapovednik	3,003.90
Katunsky zapovednik	1,500.79
Khakassky zapovednik	1251.24
Tunkinsky National Park	11,836.00
Sayano-Shushensky zapovednik	3,903.68
Ubsunurskaia Kotlovina zapovednik	396.4
	Total for hotspot 30,704.39
Far East	229,691
Bastack zapovednik	910.38
Bolon'sky zapovednik	1,036.00
Bolshekhetkhtsirsky zapovednik	451.23
Botchinsky zapovednik	2,673.80
Bureinsky zapovednik	3,584.44
Kedrovaia Pad zapovednik	179
Khankaisky zapovednik	379.89
Khingansky zapovednik	939.95
Lazovsky zapovednik	1,200.00
Sikhote-Alinsky zapovednik	3,901.84
Ussuriysky zapovednik	404.32
	Total for hotspot 15,660.85

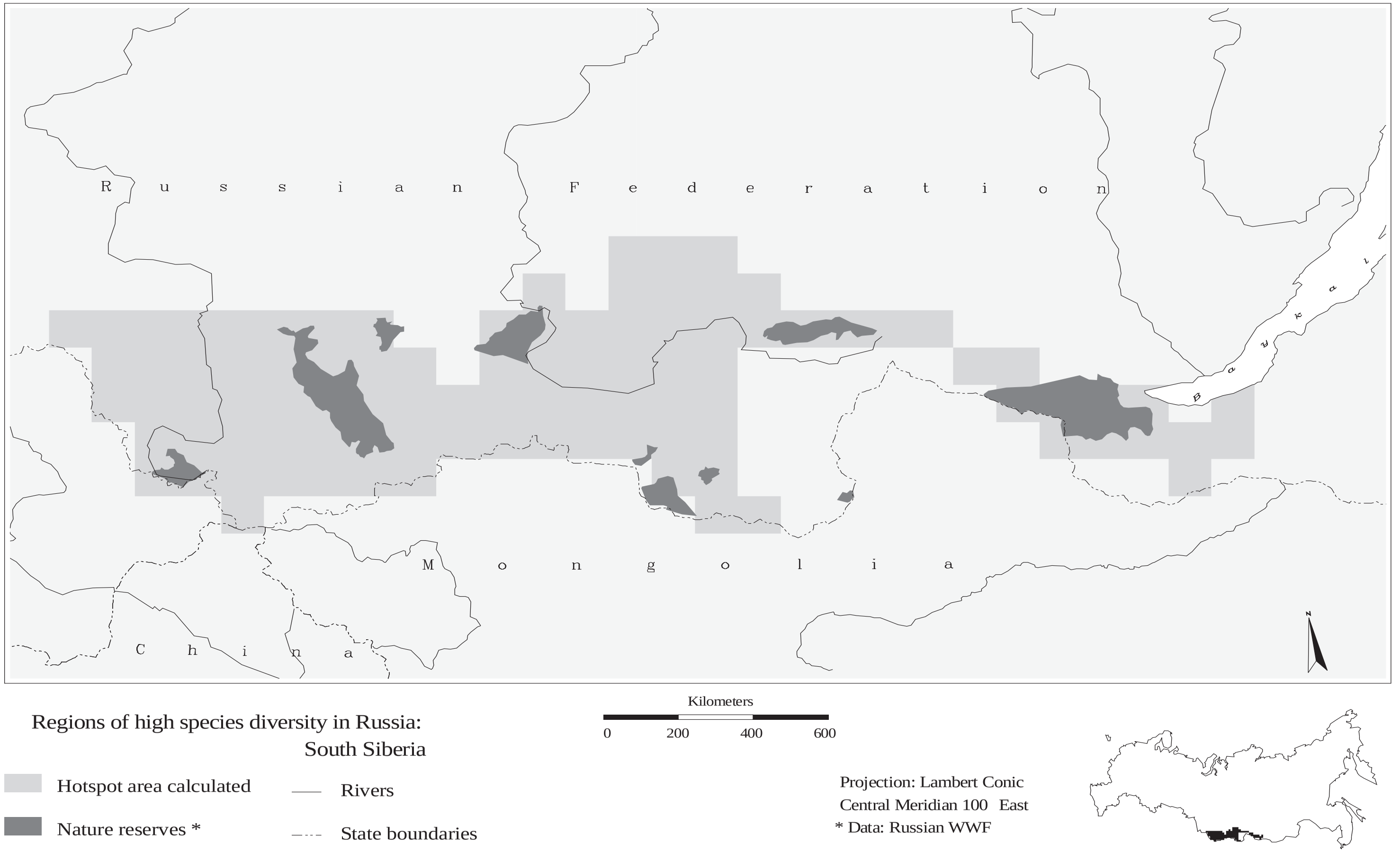
Table 1. Statutory reserves in the Russian biodiversity hotspots (Tishkov, 2002).

a)



Regions of high species diversity in Russia: North Caucasus

b)



Regions of high species diversity in Russia: South Siberia

c)

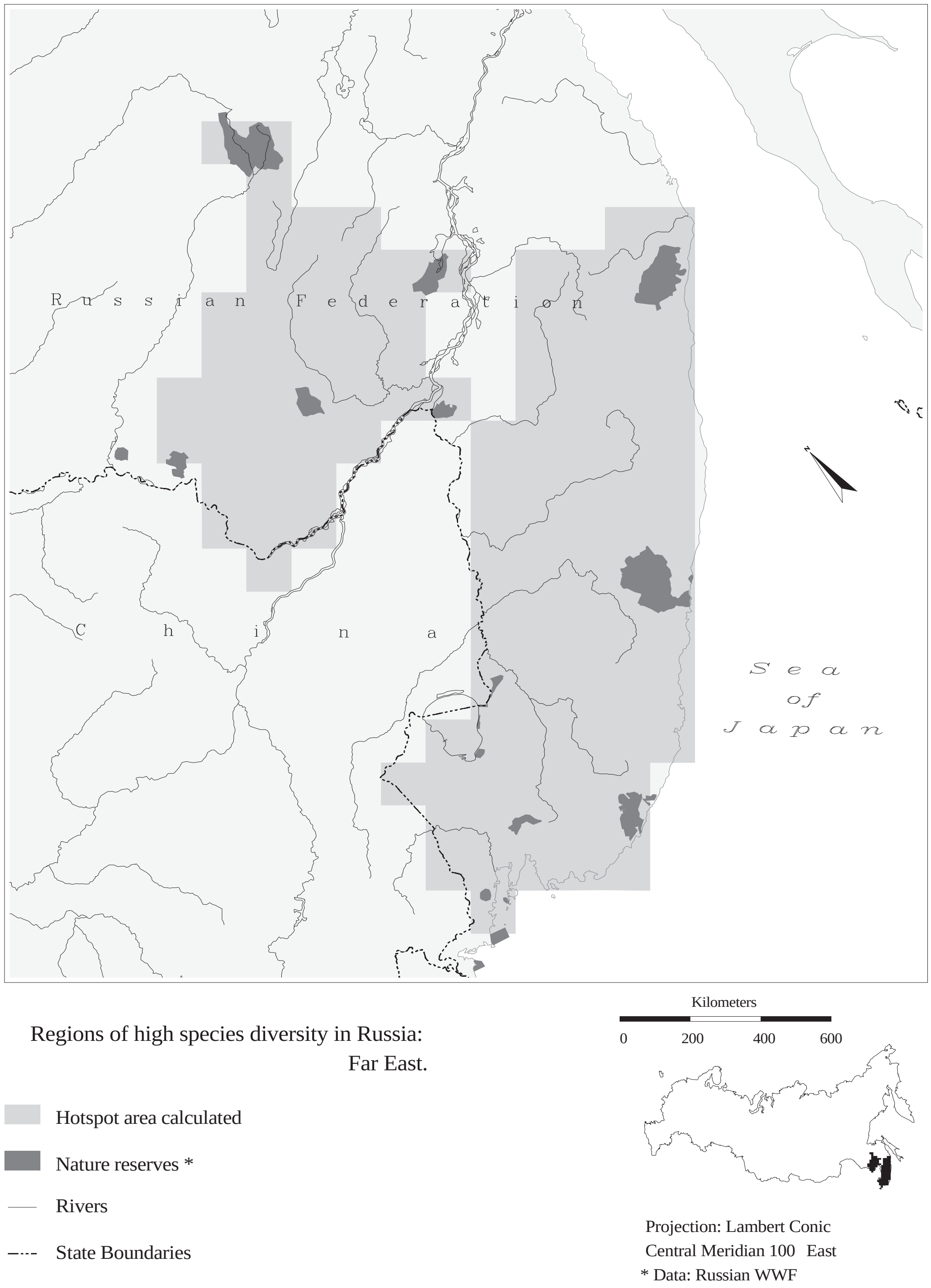


Figure 1. Statutory reserves in the three biodiversity hotspots a) North Caucasus, b) South Siberia c) Far East

The percentage of the area, designated as nature reserves in the three hotspots (see Table 2) is two to five times higher than the total share for Russia. However, the share represented by these statutory protected areas is not distributed equally between the hotspots and do so prove their relative importance in representing biodiversity. This can be seen from the example of the Red Data Book vascular plants and vertebrate animal species (reference RDB plants, animals) inhabiting the hotspots. The share of *zapovedniks* and national parks in percentage of the total hotspot area is highest for South Siberia, while the amount of the RDB species is lowest here (see Table 2).

	Share of protected areas in hotspot (percent)	Relative percentage of species from Red Data Book
North Caucasus	12	30
South Siberia	16.7	11
Far East	6.8	23

Table 2. The share of statutory nature reserves in the Russian biodiversity hotspots and the percentage of the RDB vascular plants and vertebrate animal species in the hotspots.

This discrepancy can be explained by the historical development of conservation planning in Russia. The first nature reserves in the three hotspots were established at the beginning of the twentieth century. Between the years 1916 and 1936, four nature reserves were designated in the Far East, two in North Caucasus and one in South Siberia. In the economically stable period of the former Soviet Union between the years 1960 and 1985, each of the three hotspots gained two or three new nature reserve. In the 1990 the few new nature reserves were established in the least populated and most remote regions of South Siberia, providing a greater percentage of protected area for this hotspot.

Some of the *zapovedniks* in the Russian biodiversity hotspots were recognised by UNESCO as the Biosphere Reserves (www.unesco.org/mab/): Kavkazsky and Teberdinsky in North Caucasus, Sichote-Alinsky in the Far East, Katunsky and Ubsunurskaia Kotlovina in South Siberia.

The mean share of protected areas in the Russian national biodiversity hotspots constitutes 11.4%. This value is considerably smaller than the mean share for the 25 global biodiversity hotspots (37.7% see Table 1 of Myers et al., 2000).

2.3 Distribution and environmental bias of protected areas in the hotspots

Each of the Russian biodiversity hotspots includes several vegetation zones (Venevsky and Venevskaia, 2004), ranging from mountainous tundra to coniferous or broadleaved forest and grassland areas. Such a geobotanical representation reflects the environmental gradients (climatic and edaphic), existing in the hotspots, which are caused mainly by topographic variation. Therefore an optimal reserve network in a hotspot should represent variations in relief in order to provide a variety of habitats for biological communities and species. The recent distribution of protected areas by elevation belts in the three hotspots follows historical patterns of the agricultural conquest of regions at the end of the 19th century and the development of conservation areas at later time periods.

To investigate a possible representation bias for habitats, we analysed distributions by elevation belts and altitudinal variation for all of the 0.5 x 0.5° grid cells within the three hotspots and for the cells, belonging to the protected areas.

The altitudinal location of protected areas differs according to the historical development of land use in the regions. This can be seen from comparison of the average, median, minimum and maximum values of elevation and altitudinal variation for the set of grid cells in the hotspots and in the protected areas (Table 3).

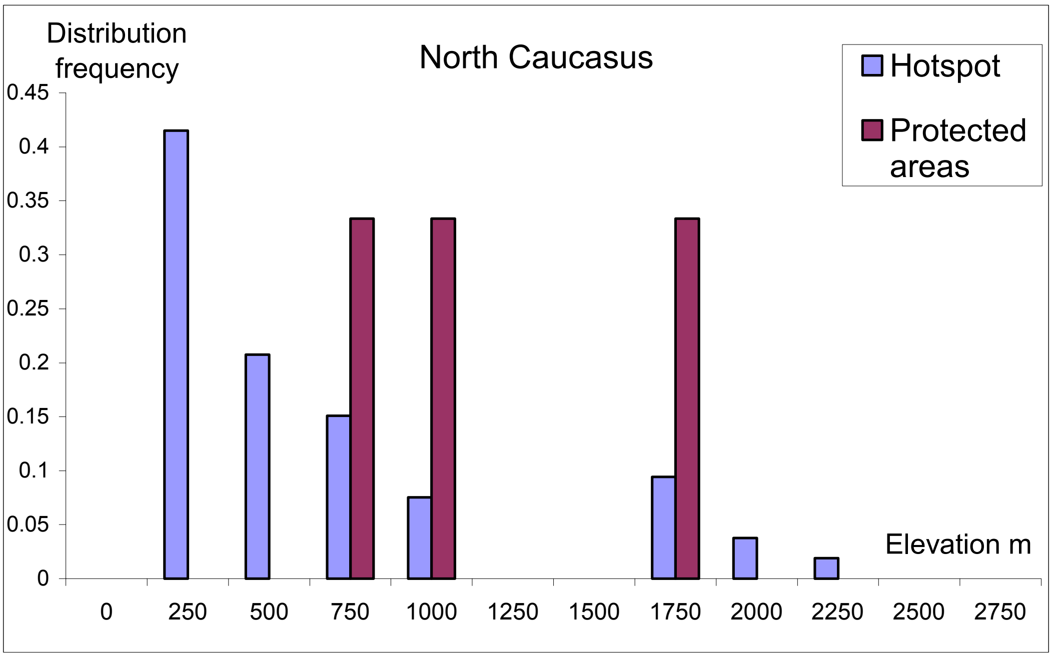
Hotspot	Elevation (m)			Altitudinal variation (m)		
		Entire hotspot	Protected areas		Entire hotspot	Protected areas
North Caucasus	Mean	543	1007	Mean	351	361
	Median	317	827	Median	314	386
	Minimum	13	537	Minimum	26	235
	Maximum	2030	1657	Maximum	837	463
South Siberia	Mean	1435	1491	Mean	230	232
	Median	1467	1383	Median	247	255
	Minimum	260	633	Minimum	2	19
	Maximum	2763	2423	Maximum	893	384
Far East	Mean	496	491	Mean	270	295
	Median	480	467	Median	182	211
	Minimum	60	60	Minimum	1	1
	Maximum	1370	1370	Maximum	990	990

Table 3. Elevation and altitudinal variation distribution parameters for the entire area of hotspots and protected areas

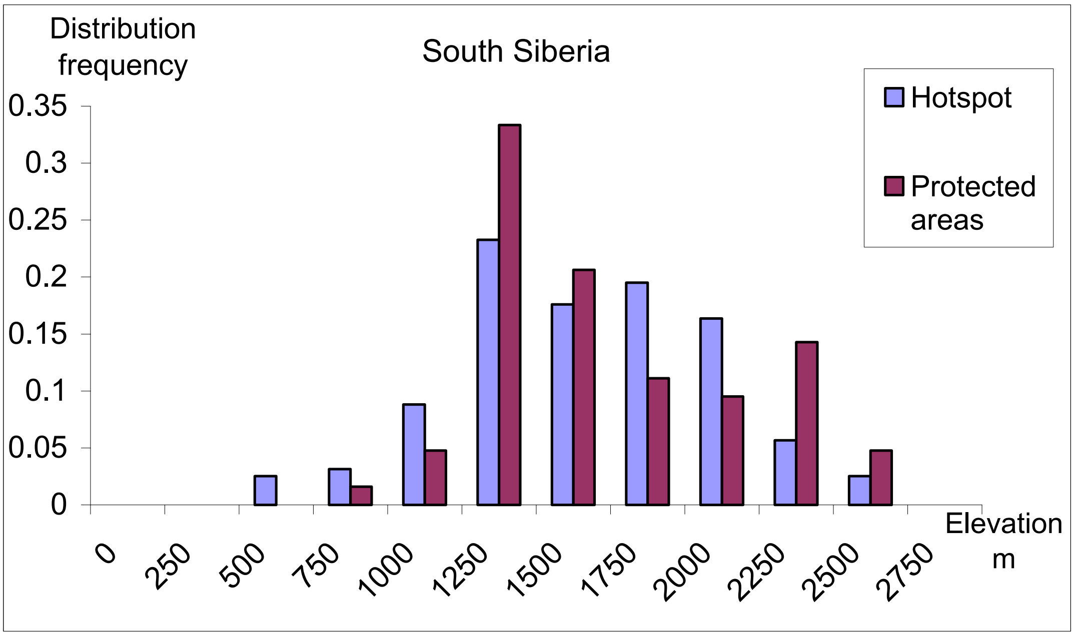
Protected areas of North Caucasus are situated in the highlands (mean elevation of protected at areas 1007 metres is almost twice that for the entire hotspot) with moderately steep slopes (variation of elevation in the grid cells of the protected areas ranges between 235 and 483 metres in comparison with the interval 26 to 837 metres for the cells of the entire hotspot). A similar, but less profound, tendency in the topographical location of protected areas can be seen in South Siberia, where the protected areas are situated in relatively flat uplands. This to some extent can be explained by the later development of agriculture in South Siberia compared with the North Caucasus. Protected areas of the Far East, which was the last region of Russian conquest in Siberia, well present both the elevation and altitudinal variation in the entire hotspot (Table 3).

Comparison of the frequency distributions by elevation of grid cells for the entire hotspots and for their protected areas proves that statutory reserves of North Caucasus have the largest environmental bias (Figure 2). Indeed, the frequency distribution by elevation for the cells of this hotspot differs radically from the distribution for protected areas (Figure 2a). The protected areas of the Far East are not biased by elevation (Figure 2c), while nature reserves in South Siberia have moderate overrepresentation of the elevation belt between 1000 and 1500 metres (Figure 2b)

a)


North Caucasus
Distribution frequency
0.45
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
0
250
500
750
1000
1250
1500
1750
2000
2250
2500
2750
Elevation m
Hotspot
Protected areas

b)


South Siberia
Distribution frequency
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
0
250
500
750
1000
1250
1500
1750
2000
2250
2500
2750
Elevation m
Hotspot
Protected areas

c)

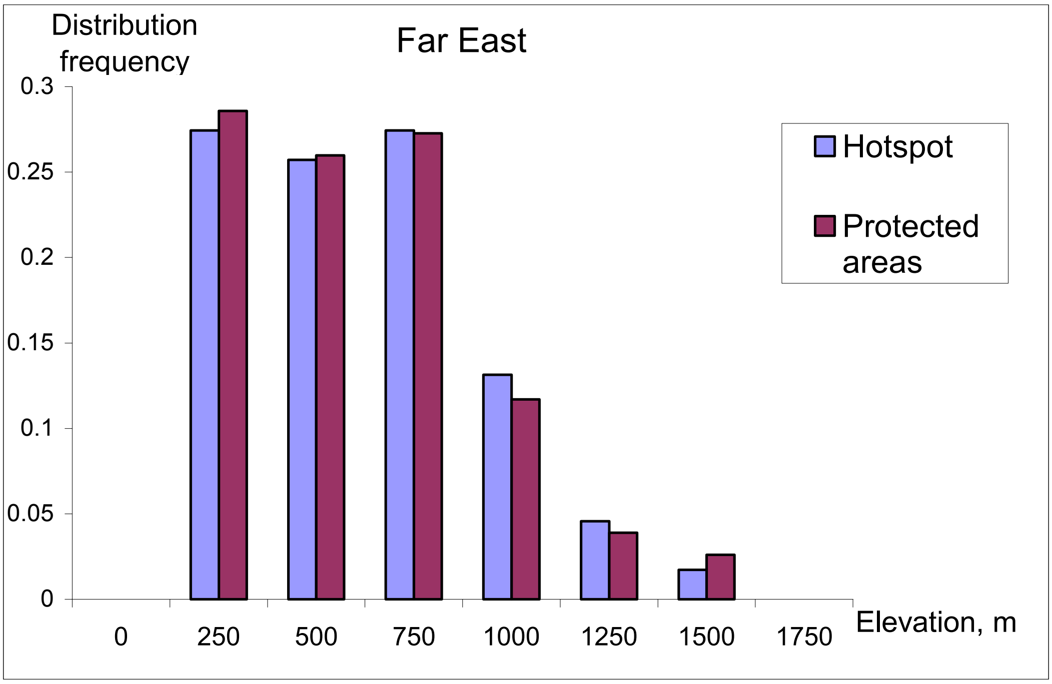


Figure 2. Frequency distribution by elevation for the grid cells of a hotspot and its protected areas, a) North Caucasus, b) South Siberia, c) Far East.

The long-term persistence of species in the hotspots can be provided by protection of differential habitat areas, in particular differential elevation belts. Therefore, we conclude that the North Caucasus biodiversity hotspot requires substantial additional area for new nature reserves.

2.4 Efficiency of species representation by protected areas in the hotspots

Several measures were suggested to evaluate the performance of a reserve network in a region. For instance, the efficiency of a reserve network is inversely proportional to its relative cost, described as occupied percentage of area in the region (Pressey and Nikholls, 1989). Effectiveness can be estimated by the percentage of the overall number of species found in protected areas - the most traditional measure - or by the relative representation of the restricted range species (Rodrigues et al., 2000). These measures may be also corrected by considering only species in local peaks of their abundance (Gaston and Rodrigues, 2003).
Where quantitative conservation targets have been determined, the effectiveness of a reserve network can be determined by the percentage of fulfillment of these targets (Cowling et al., 2003), the approach we are going to adopt in this study.
Our conservation targets for species found in the three Russian biodiversity hotspots are presented in terms of the number of locations or size of area necessary for the persistence and retention of the species (Venevskaia and Venevsky , 2004). Species in the Red Data Book (RDB) in the five taxa used in global biodiversity hotspots analysis (terrestrial mammals, amphibian, reptiles, birds and vascular plants), were selected in order to set the quantitative conservation targets (the complete list of the targets can be found at www.pik-potsdam.de/~irina/targets). We elaborated targets (baseline and retention), reflecting differential requirements for regional protection, caused by the regional nature of the

community structure and the specific human threats for 288 RDB species, found in the five groups in the national hotspots (Table 4).

Species from the Red Data book	North Caucasus	South Siberia	Far East
Vascular plants	86	20	57
Birds	24	22	37
Mammals	14	8	8
Reptiles	8	0	1
Amphibian	2	0	1
Total	134	50	104

Table 4. Rare, threatened and endangered species in the three hotspots for which the quantitative conservation targets were set (Venevskaia and Venevsky , 2004).

In general, an assessment of the relative performance of existing statutory reserves in the hotspots can be achieved by averaging of the percentages of fulfillment of the conservation target for each species. However, the application of simple averaging might result in obscuring the variation in fulfillment of conservation targets across the RDB species, for example, when significant part of a species population are located outside a reserve network, while other are situated completely inside protected areas. To avoid this situation, we applied not only the percentage of fulfillment of the conservation target averaged across the species, but also its median value.

Such an assessment allows direct comparison of the ability of the existing reserve networks in the hotspots to protect different taxa in view of our conservation targets.

The reserve networks in the three hotspots have a similar distribution of the percentage of fulfillment of the conservation targets by taxa, if values averaged for the species are used (Table 5). Vascular plants are the most protected taxa, mammals are poorly protected, while birds are somewhere in between. Reptiles and amphibian species are almost absent in the protected areas (the average percentage of fulfillment of conservation targets is 1% and 2% for the Far East and North Caucasus).

	Vascular plants	Birds	Mammals
North Caucasus			
Average fulfillment of conservation targets (%)	49	27	14
Median fulfillment of conservation targets (%)	25	10	9
South Siberia			
Average fulfillment of conservation targets (%)	57	28	19
Median fulfillment of conservation targets (%)	50	11	8
Far East			
Average fulfillment of conservation targets (%)	31	23	22
Median fulfillment of conservation targets (%)	18	4	14

Table 5. The percentage of fulfillment of conservation targets in the hotspots' nature reserves by taxa

Comparison of the median percentage indicates some variations from this general rule. It can be seen that a considerable amount of the RDB vascular plant species in North Caucasus and Far East and the majority of the RDB bird species in the Far East are not present in the reserve networks of the regions.

The percentage of fulfilment averaged over four group of species (vascular plants, birds, amphibian/reptiles and mammals) is similar for all the hotspots:

- 23% for North Caucasus;
- 34% for South Siberia;
- 19% for the Far East.

Not more than one third of the conservation targets set for the Russian biodiversity hotspots is achieved within the existing system of statutory reserves. This means that additional protected areas are necessary and should be designed in view of our conservation targets.

We suggest measuring the efficiency of the recent reserve networks by the percentage of fulfilment of conservation targets averaged by the taxa, divided into the percentage of the protected area in the hotspots. Such a definition of the efficiency well represents the minimum area requirement and can be calculated either by the achievement of conservation target averaged across a species or by its median value across the species.

The simple efficiency analysis demonstrates an advantage of the reserve networks in the Far East in comparison with the other two regions, which have the similar efficiency values despite considerable differences in the total protected area (see first column of Table 6).

	Efficiency average	Efficiency median
North Caucasus	1.9	0.9
South Siberia	2,1	1,3
Far East	2,8	1,3

Table 6. Efficiency of the conservation targets achievement by the biodiversity hotspots

3. An integrated conservation plan for Russian biodiversity hotspots

3.1 Data sets for conservation planning

The data for biodiversity patterns in the three hotspots comprised the digitised distribution areas for 288 species from the Russian Red Book in the five taxa: terrestrial mammals, birds, amphibian, reptiles and vascular plants. The data (available at www.pik-potsdam.de/~irina/distributions) are presented in the ARC-INFO vector format for areas of the animal species and in the point format for the locations of plant populations. The accuracy of the digitalisation is limited by the graphical quality of the source maps in the Red Data Book (Red Data Book of the Russian Federation / Animals, 1983; Red Data Book of Russian Federation / Plants, 1984) and is not less than the visually distinguishable distance between the two closest lines on the coarsest map for the areas of 288 species. This coarsest map in the Russian Red Book is given for one of the North Caucasian plant species. The 8 km distance between two close rivers, the rivers Terek and Kuban, is still visible on this map, so the accuracy for the biodiversity pattern data is not less then this value.

The RDB species abundance, presented in the data, may also have changed since the time the Russian Red Data Book appeared, making a slight overestimation of the distribution areas for the species possible.

The geographical borders of the three biodiversity hotspots were determined at the 0.5º x 0.5º spatial resolution (Venevsky and Venevskaia, 2004). These borders were slightly modified in this study by exact matching to the state border of Russia.
The map of Russian statutory reserves (*zapovedniks* and national parks) in vector GIS format was prepared by the Russian WWF (Ofrenia, personal communication)

3.2 Planning unit layer

As a rule, a spatially explicit conservation plan is based upon *a priori* subdivisions of the landscape according to planning units (Pressey and Logan, 1998). Experts or automatic reserve selection procedures assess the importance of each planning units for achieving quantitative conservation targets.
For the Russian biodiversity hotspots we developed a subdivision of the landscape, which includes statutory reserves and arbitrary grid cells. These grid cells were planning units with a size of 0.125º x 0.125º (approximately 6400 ha). This size of planning unit, determined by the accuracy of our digitalisation, gave a number of units, that is appropriate for broad-scale planning, but which may not be sufficient for fine-scale implementation of conservation measures (see Section 5). A similar size of planning units (3900 ha) was successfully applied for a large-scale systematic conservation planning in the Cape Floristic Region, South Africa (Cowling and Pressey, 2003). In order to investigate the contribution of statutory reserves to target achievement we used the exact borders of protected areas. The habitat areas of rare vertebrate animals were defined within the planning units by their exact borders at 0.125º x 0.125º resolution and the point locations of the rare plants were prescribed to appropriate 0.125º x 0.125º grid cells.
The total planning unit layer for the three biodiversity hotspots consisted of the 25 statutory reserves (12% of the total area of the three hotspots) and 5424 grid cells (88% of the area).
Each of the 5449 selection units was characterised according to the number of plants populations of the RDB species, areas of the RDB vertebrate animal species in each of the three groups (mammals, birds and amphibian/reptiles) and proximity to the nearest statutory reserve measured from the geometric centre for the grid cells.

3.3 Planning procedure

3.3.1. Aim of conservation planning

In developing conservation plans, experts in North America (Groves et al., 2000) and in the Southern hemisphere (Margules and Pressey, 2000) identify as a main aim the achievement of targets for all biodiversity features. This aim allows of an outcome to be presented independently of the socio-economic situation in a region. As an aim of our conservation plan, however we suggest achieving 100% of the identified targets (Venevskaia and Venevsky , 2004) for the RDB species in only one of the four groups (vascular plants, amphibian/reptiles, mammals, birds), which we name an umbrella group for a hotspot. This aim is combined with requirements to maximise the percentage of target achievement for the other groups and to minimise the prospective conservation area. The rationale here is the design of a national large-scale conservation plan which does not pose too unrealistic area requirements and allows further fine-scale development with orientation mainly upon one group of species in a region. This represent a mid-way approach between those national conservation plans designed on the basis of biodiversity data for the single taxa (e.g. birds in Ecuador (Sierra et al., 2002) and national conservation plans which present as many biodiversity features as possible, but without area limitations (e.g. CAPE plan, Cowling et al., 2003). The identification of an umbrella group for RDB species in a hotspot also narrows the group of experts necessary to further refine regional conservation plans (see Section 5).

Our conservation plan assumes the use of formal algorithms to identify prospective conservation areas, similar to other studies adopting a systematic approach (Lombard et al., 1997; Howard et al., 1998; Pressey and Cowling, 2001). Our approach, however, is not polar to the expert-driven identification of priority areas for conservation (Hannah et al., 1998; Rodriguez and Young, 2000). Indeed, both the Red Data Book and the data on global biodiversity hotspots used to determine the borders of our three regions (Venevsky and Venevskaia, 2004) and for the design of the quantitative conservation targets (Venevskaia and Venevsky , 2004), already contain a considerable amount of expert knowledge. The use of quantitative algorithms to identify prospective conservation areas at the national scale has certain advantages: use of consistent quantitative data, flexibility to change both data and targets, and the opportunity to assess the options for achieving targets.

3.3.2. Planning protocol

The quantitative optimisation criteria for our planning protocol have two components: to maximise the percentage of fulfillment for conservation targets averaged across the five groups of species (vascular plants, amphibian/ reptiles, birds and mammals) while minimising the area required. These two components were combined in the efficiency change from the value observed for recent protected area networks (see Section 2.4). While basic optimisation is performed for the percentage of fulfillment of conservation targets averaged across the RDB species and the efficiency change based on it (see Section 2.4), possible negative effects of such averaging are neglected by the simultaneous analysis of the median values in the optimisation criteria.

We adopted the existing set of statutory reserves in the three Russian biodiversity hotspots as a fixed feature in our planning procedure. We incorporated the statutory reserves into the plan because considerable establishment, management and research investment has been already made.

The planning procedure has two possible paths: to take the vascular plants species as the umbrella group in optimisation; or to take the bird, mammal and amphibian/reptiles species as the group for which 100% of conservation targets should be achieved. We unified the RDB amphibian and reptiles species into one umbrella group, because they have environmentally complimentary habitats (dry and wet) and they are small in numbers (Table 4).

If the umbrella group comprises the vascular plant species, then the selection procedure has following steps:

Step 1. Vascular plant species with all locations of populations identified as a conservation target are listed. The set of planning units (grid cells) entirely containing the locations of these species and the areas of statutory reserves provide an initial large-scale conservation area.

Step 2. The percentage of conservation targets achieved is calculated for the RDB vascular plant species of the umbrella group and sorted in ascending order. The planning selection unit (grid cell) with the minimum distance from the borders of the already designated conservation area and with the maximum percentage of conservation target fulfillment for the first species in the list is found and is included in the new conservation area. When several choices for such a planning unit are possible, the one increasing the entire contingency of a conservation area is chosen.

The second step is iteratively repeated until one hundred percent of conservation targets are achieved for the RDB vascular plant species.

Where the umbrella group is mammals, birds or amphibian/reptiles, the selection procedure has the following steps:

Step 1. Species from the umbrella group with the entire area of distribution as a conservation target are listed (these species comprise more than half the RDB vertebrate animal species found in the regions). Their distribution areas and the areas of statutory reserves provide the initial contours of a conservation area.

Step 2. The percentage of conservation targets achieved is calculated for the RDB species of the umbrella group and sorted in ascending order. The planning selection unit (grid cell) with the minimum distance from the borders of the conservation area from a previous step and with the maximum percentage of conservation target fulfillment for the first species in the list is found and is included in the new conservation area without consideration of contingency between grid cells.

The second step is iteratively repeated until one hundred percent of conservation targets are achieved for the umbrella set of the vertebrate species.

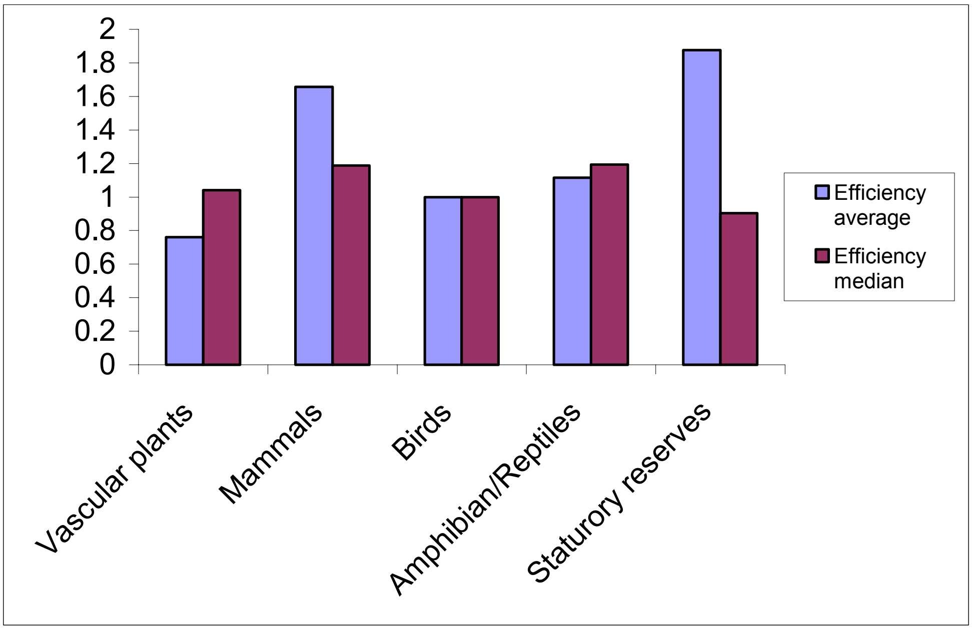
The selection procedure results in three variants for conservation areas for the South Siberian hotspot and the four variants for the other biodiversity hotspots.

The final variant for a conservation area in a hotspot is decided by our optimisation criteria, which is measured by the change from the *de facto* efficiency value for the existing statutory reserves. A prospective conservation area for an umbrella group of species with the minimum (for average value across species) and maximum (for median value across species) efficiency change provides a regional conservation plan. The simultaneous calculation of the two variants of optimisation criteria, based on median for the species values and average for the species values of the percentage of conservation target fulfillment, allows situations where species are considered asymmetrically to be avoided.

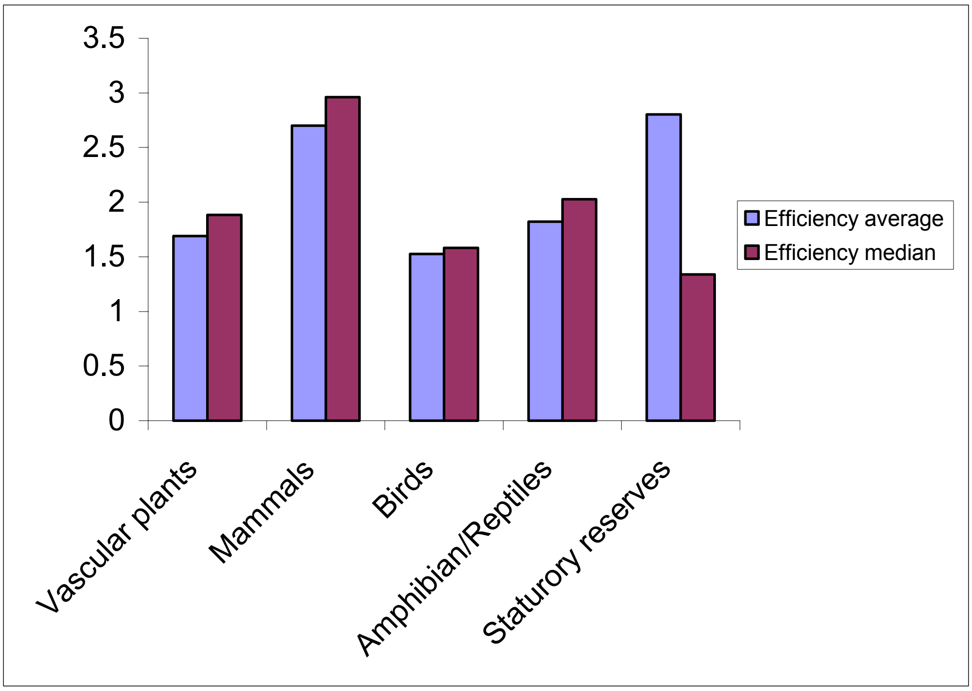
3.4 Results

The results of the planning procedure for the biodiversity hotspots can be summarised from the variation in efficiency calculated for prospective conservation areas of the four (three in South Siberia) umbrella group of species (Figure 3).

a)



b)



c)

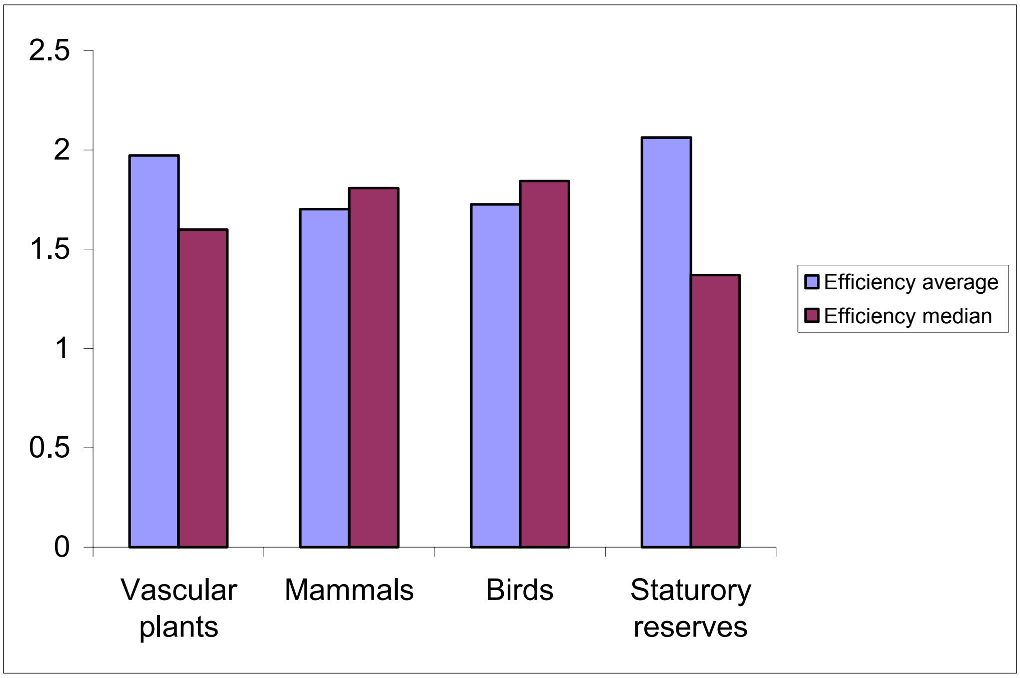


Figure 3. Efficiency in achieving conservation targets for the umbrella groups of species for the three Russian biodiversity hotspots a) North Caucasus b) Far East and c) South Siberia.

Despite regional differences in the distribution of efficiency in the achievement of conservation targets, one can see common trends for all the hotspots. Indeed, the efficiency calculated by averaging across the RDB species of the percentage of conservation targets fulfilment for all variants of the new conservation areas is generally lower (although sometimes marginally) then the efficiency for the existing reserve networks. However, the efficiency calculated using the median for the RDB species values for the new variants is always larger then the present one.

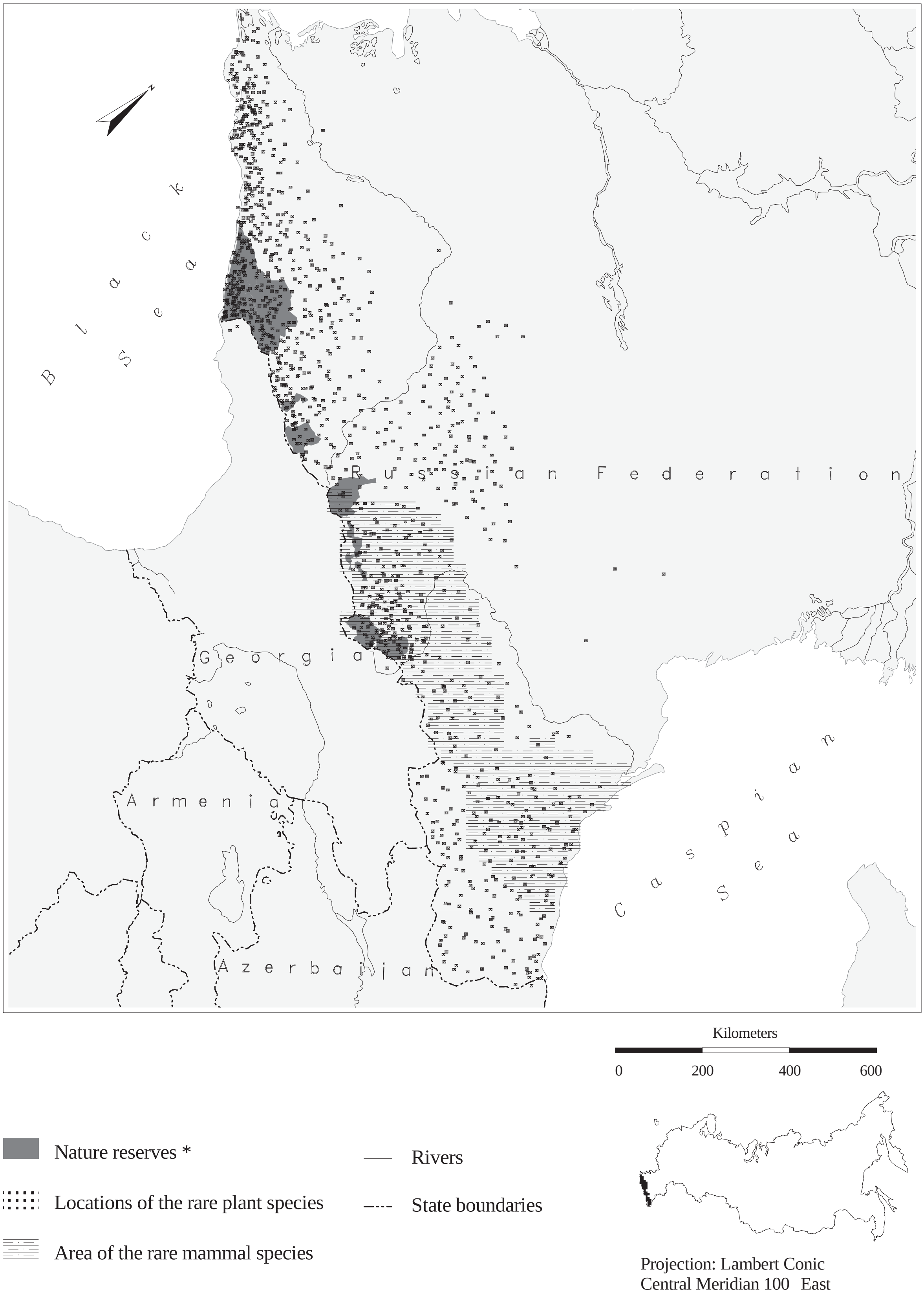
Both an increase in the area and the percentage of achievement of conservation targets achievement affect the value of efficiency during the selection procedure. This means particularly that changes in the optimisation criteria sometimes may require accurate interpretation with analysis of other quantitative characteristics for priority areas.

3.4.1 North Caucasus

North Caucasus has the system of statutory reserves, which is the most environmentally biased (see Section 2.3) and is the least efficient in the sense of our conservation targets (see section 2.4).

Application of the algorithmic selection procedure showed significant areal demands for new protective areas in North Caucasus. For instance, the entire area of the hotspot is required for the achievement all the conservation targets for the RDB bird species.

A change in the efficiency of conservation areas for four umbrella group demonstrates a better performance for the plans designed for mammals and amphibian/reptiles as the 100% targeted group. This can be seen from the minimum change of the averaged efficiency and the maximum change of the efficiency median for these two variants of conservation areas (Figure 3a). However, the conservation plan, elaborated for mammals as the umbrella species group provides only approximately 40% achievement of all conservation targets and less then half of conservation targets for the vascular plant species, because the areas of many RDB vascular plant species located outside of the area of the RDB mammal species (Figure 4).



Projection: Lambert Conic
Central Meridian 100 East

* Data: Russian WWF

Figure 4. The area of RDB mammal species and point locations of the RDB vascular plant species in the North Caucasus

The conservation plan in which the amphibian/reptiles species is the umbrella group fulfils 92% of the conservation targets for the RDB species of North Caucasus and has the best optimisation criteria values (Figure 3a) and minimum area in comparison with the other variants of prospective conservation areas. Therefore, despite the large area demand (82% of the biodiversity hotpot), this variant is suggested as the large-scale regional conservation plan (Figure 5).

Figure 5. Large scale conservation plan for the Russian national biodiversity hotspot North Caucasus

3.4.2 Far East

The conservation plan with the RDB mammal species as the umbrella group has clear advantages in the value of the optimisation criteria over that of other variants for the Far East (Figure 3b). Indeed, this plan requires only 31% of the hotspot area and provides achievement of 81% of the conservation targets for the entire set of species (Figure 6)

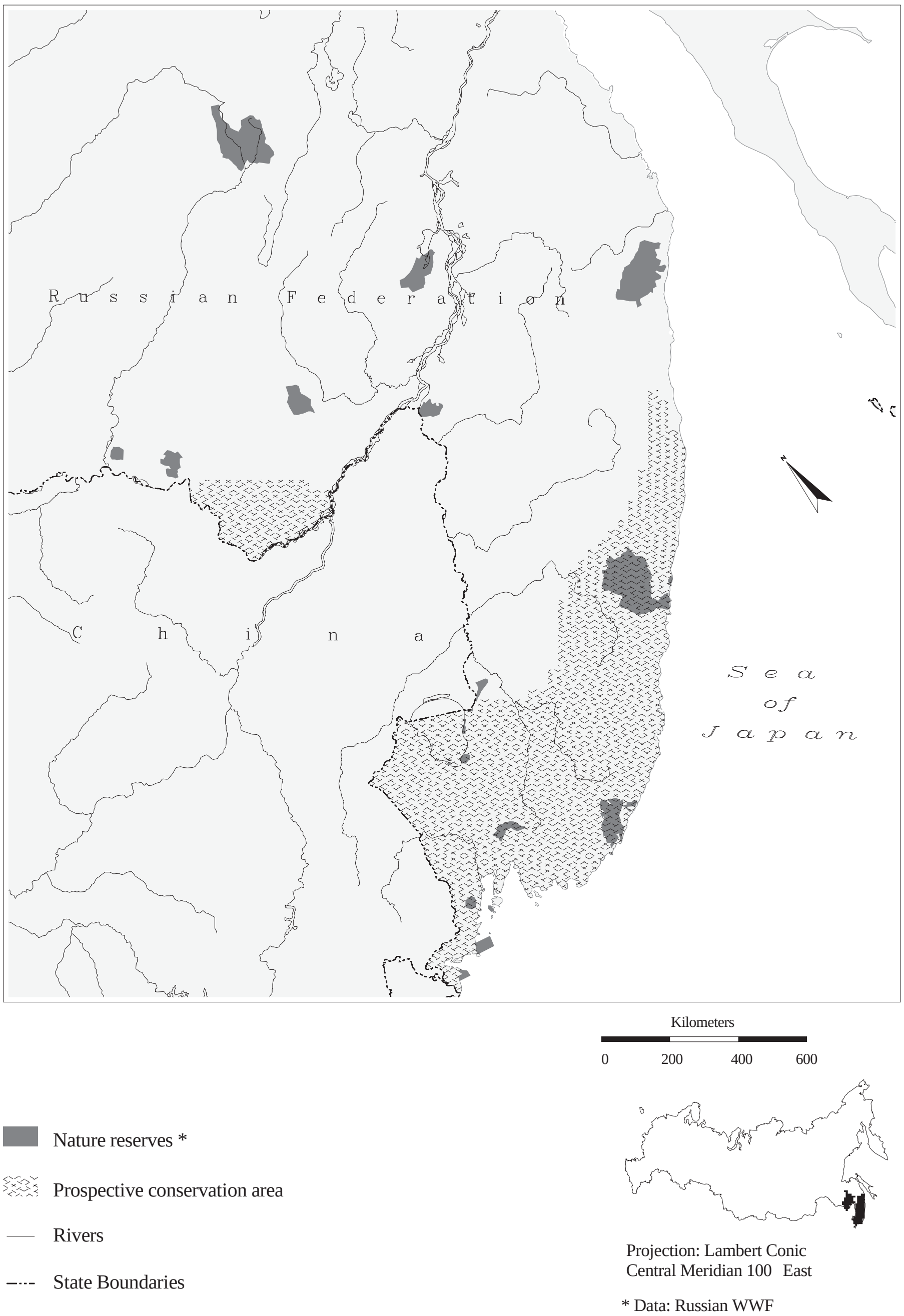


Figure 6. Large-scale conservation plan for the Russian national biodiversity hotspot Far East

3.4.3 South Siberia

The areas of the RDB vascular plant species and the areas of the RDB mammal species do not coincide well in South Siberia, similarly to in the North Caucasus (Figure 7).

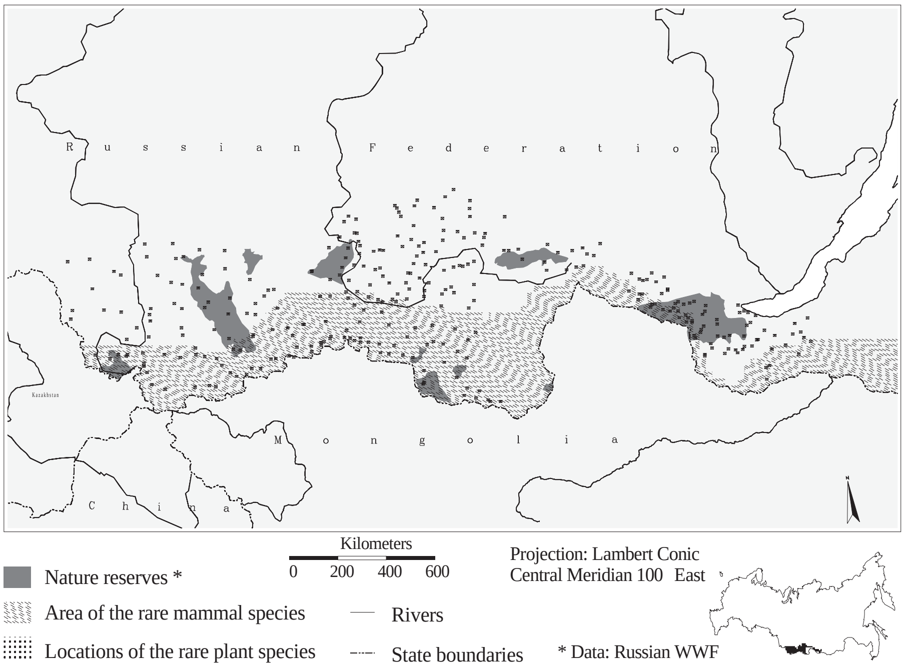


Figure 7. The area of RDB mammal species and point locations of the RDB vascular plant species in South Siberia

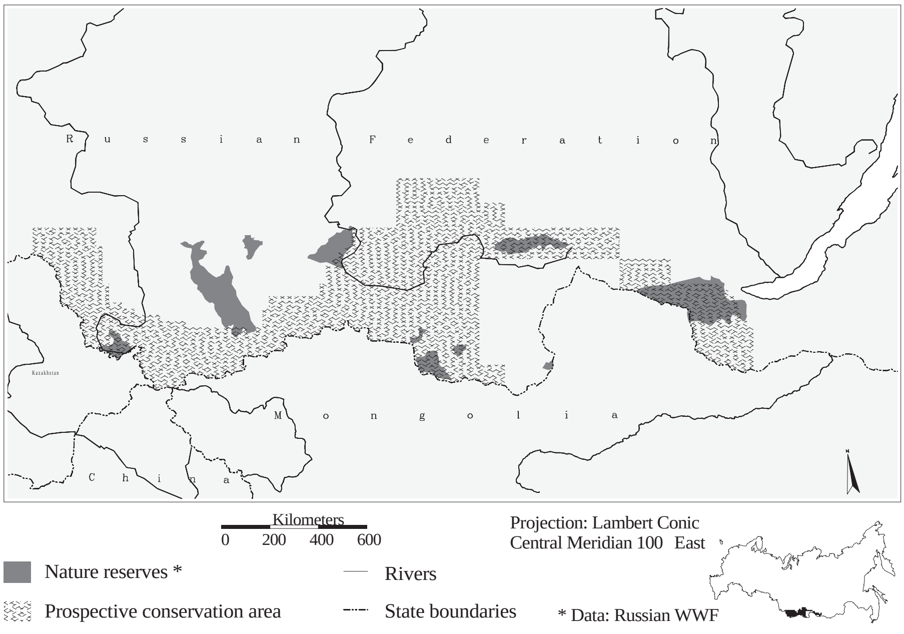


Figure 8. Large-scale conservation plan for the Russian national biodiversity hotspot South Siberia

So the conservation plan, which uses the mammal species as the umbrella group, does not cover many localities of the rare plants populations. The total percentage of conservation targets fulfillment is 75% for this variant.

We designed the regional conservation plan for South Siberia (Figure 8), based on the requirement of total achievement of conservation targets for the RDB bird species. This variant results in a slightly better optimisation criteria value in comparison with the other plans and provides 85% of conservation target achievement (Figure 8).

3.4.4 Inter-comparison of the regional conservation plan

The design of conservation plans for the Russian biodiversity hotspots confirmed that no certain taxonomic group of species can be preferred in the elaboration of a national conservation plan. Indeed, the umbrella groups of the RDB species which provide the best conservation plans, are taxonomically different for the three hotpots. This means that national conservation plans based on a single charismatic group of species may not provide an optimal conservation strategy.

The requirements for priority areas differ significantly across the biodiversity hotspots, but can be seen from the average efficiency values (Table 6). Indeed, the largest ratio for a prospective conservation area, 82%, is required for North Caucasus, South Siberia follows at 49% and less than a third of the hotspot area is required for the large-scale conservation plan for the Far East.

4. Summary of hierarchical conservation plan at national level for Russia

The suggested large-scale national conservation plan for Russia includes four analysis steps for systematic conservation planning (Margules and Pressey, 2000): analysing and mapping of biodiversity features (Venevsky and Venevskaia, 2004), elaboration of quantitative conservation targets (Venevskaia and Venevsky , 2004), review of existing reserve networks (see Section 2) and design of new conservation area (see Section 3). Our hierarchical approach adopts as widely novel elements of ecoregional conservation planning (e.g. algorithms for determining the borders of national biodiversity hotspots, regional inter-comparison of the hotspots in view of conservation targets). The new conservation areas are designed by a formal optimisation procedure, based on GIS data for biodiversity features and human threats patterns, i.e. by approach widely used in contemporary conservation planning (Araujo, 1999; Cowling et al.2003).

Our large-scale conservation plan combines the advantages of a systematic approach and expert knowledge, for instance, with Red Data book information on anthropogenic and natural threats for the rare, threatened and endangered species. We are however, aware that the data on biodiversity features used in the study is rather limited and is likely to have some gaps which should be identifeid by additional expert analysis. We suggest this large-scale conservation plan as a starting point for further consideration by field experts and practical managers, who can conduct further selection and implementation of reserve networks within the suggested conservation areas in the three regions at fine-scale spatial resolution. A supporting fine-scale expert-driven study can bring additional expert judgements about biodiversity persistence and implementation options (i.e. recent economic constraints of implementation and possible rationalisation of existing statutory and non-statutory reserves), which were not included in our study. Indeed, the comparison study for priority conservation areas selected by experts and reserve optimisation software in the Cape Floristic Province demonstrated that preliminary exposition to systematic conservation planning outputs is beneficial for experts and helpful for the design of their final wishlist areas (Cowling et al., 2003).

5. Implementation issues

Not many studies, published in the literature, have considered issues related to conservation plans being implemented in reality (but Cowling and Pressey, 2003). Below we discuss these issues in outline in relation to the recent situation in Russia.

5.1 Scheduling

Despite the relatively low cumulative ratio of the identified priority area to the total area of Russia, equal to 1.28%, significant land resources are needed to implement the suggested large-scale national conservation plan. Indeed, the required area additional to the existing statutory reserves for the three hotspots is 167 400 km^2, which would quadruple the area under some form of protection in the three biodiversity hotspots. Therefore, the nation wide reservation of land resources should be scheduled according to a realistic and efficient strategy, setting priorities in the timing of conservation measures within the hotspot and between them.

One such scheduling strategy assumes focusing initially on areas of high biodiversity value and high vulnerability to human threats. Taking this strategy for Russia means applying conservation measures especially within the North Caucasus hotspot, where the existing reserve network is the most biased environmentally, least efficient, and habitats are highly fragmented due to anthropogenic pressure, such that its additional area requirements are the largest of the three regions.

Another alternative view of scheduling is to reserve initially large areas of intact habitats, where it is feasible to accommodate a considerable amount of biodiversity features. Such a strategy can provide greater biodiversity returns per unit of management investment (Ferrier, 2002; Noss et al., 2002). The conservation areas in South Siberia, where the land use ratio is lowest should be developed initially according to this strategy.

We, however, advocate a scheduling strategy, which prioritises the hotspots according to the opportunity profit, calculated as the efficiency increase per unit of area in relative units. Thus, we suggest initially developing the reserve networks of the Far East, where the relative opportunity profit is the highest (see Figure 9).

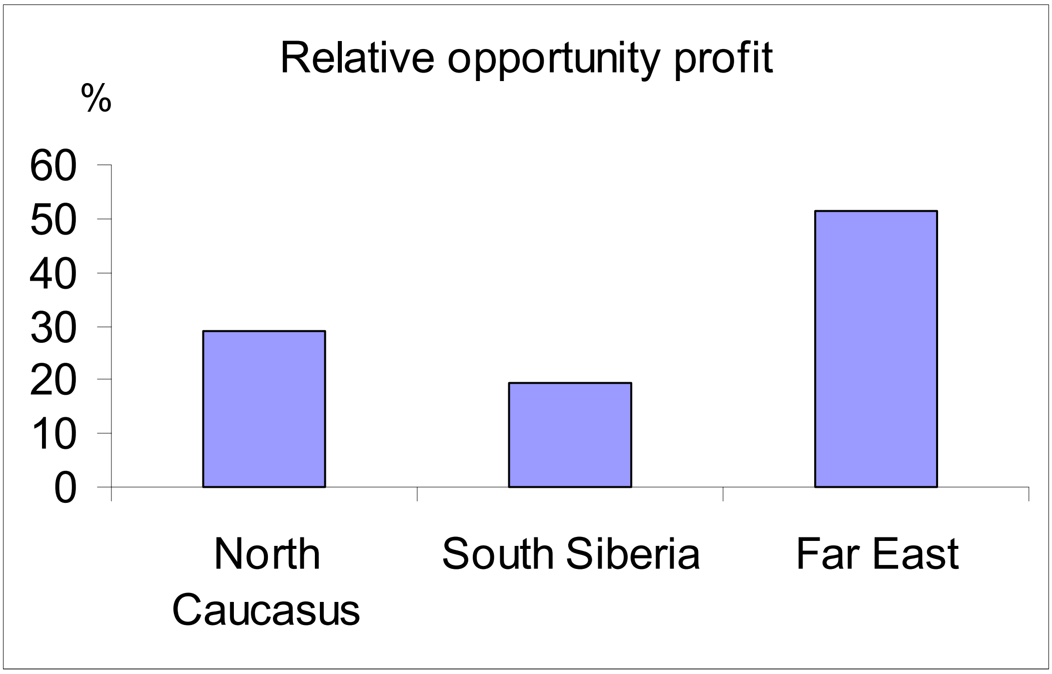


Figure 9. Opportunity profit in land reservation for the three Russian biodiversity hotspots, calculated as the median efficiency increase per area unit in the regions (normalised to 100%).

We suggest accepting such a prioritisation scheme in scheduling as well, because the North Caucasus region today, needing the development of new protected areas the most badly, is prone to political and military conflicts, causing the danger that the results of conservation actions will be neglected.

5.2 Forms of protection

The large-scale conservation plan assumes a massive increase of protected areas outside the existing reserves. It is, probably not feasible for Russia, a country with a transition economy, to pay the high opportunity costs and include a major part of the targeted lands in the statutory reserve system.

Enhancing the legislative status of the existing natural reserves and hunting grounds (*zakazniks*) of local and regional jurisdiction in the hotspots may provide a major support in implementing of our conservation plan in reality. Another challenge is using off-reserve mechanisms for conservation in the three hotspots. These mechanisms include providing financial incentives for landowners to set aside some of their properties for conservation purposes, and changing the management of agricultural landscapes, especially in North Caucasus, to biodiversity-friendly options.

We do not, however, exclude the possibility of a fast increase in conservation status for some of the existing non-statutory reserves. Indeed, the federal natural *zakazniks* in the hotspots have a significant potential (land recourses and management staff) for conservation of biodiversity. They could comprise a basis for future statutory reserves in the hotspot. The nine federal *zakazniks* of North Caucasus have a total area of 3 532 km^2 , and the five in the Far East cover 8 862 km^2 , while in Southern Siberia the three *zakazniks* occupy 2 900 km^2 (www,biodat.ru). This distribution of potential areas for new statutory reserves is completely in line with the suggested scheduling strategy, giving top priority to the Far East, where approximately one quarter of the required additional conservation area can be covered by the existing federal *zakazniks*. The feasibility and possibility of the transition from federal *zakazniks* to statutory reserves can be studied during fine-scale regional conservation planning in the three hotspots with the participation of stakeholders in the regions.

5.3 Costs

The costs of conserving the areas identified in the three Russian biodiversity hotspots should be accurately estimated at the stage of fine-scale regional conservation planning. The land acquisition, management and transformation in status of protected areas in the 20-year implementation of the conservation plan for the Cape Floristic Province were estimated at $75 million per year (Frazee et al., 2003). Because the additional area of 1.5 million ha allocated for strict reservation in this plan is similar to our value, one can expect similar costs for Russia.

These costs should be derived both from federal and local sources, and would most likely fall mainly on the shoulders of the nature protection agencies of the highest Russian administrative units (i.e. *oblast'*s, *krai*s and republics) in the hotspots.

5.4 Ongoing review and refining of the large-scale conservation plan

Our formalised approach allows a periodical review of the large-scale conservation plan for Russia. It must be updated as new data on biodiversity features (e.g. patterns of RDB species) becomes available or new conservation targets emerge or have to be adjusted.

The suggested large-scale plan relies on further detailed refining at a landscape level. The identified priority areas should be additionally studied by experts and land managers, who can elaborate detailed fine-scale conservation plans and implementation schemes for reserve and off-reserve conservation mechanisms. The existence of umbrella groups of the RDB species, recognised by our conservation plan, in the three biodiversity hotspots makes it easier to select regional experts for detailed conservation planning (for example, researchers and species managers for mammals in the Far East can substantially contribute to the regional fine-scale conservation plan).

6. General conclusions

In our three papers with the common name "Hierarchical systematic conservation planning for a national level: example of Russia" we attempted to elaborate a quantitative hierarchical methodology, which we believe can be applied generally in many large countries. It requires as an input large-scale climate data, which is in public domain, and the national Red Book data, which is often available. As the output, our methodology provides a national conservation plan with a large-scale spatial resolution, determined by available basic data. Such large-scale plans can provide a basis for detailed regional conservation programmes in large countries.

We are, however, aware that we make many *a priori* hypotheses in our approach, which may limit its application in certain countries. There are probably no clear remedies in large-scale national conservation planning that are applicable for all countries, and the feasibility for the application of our approach should be always studied in each separate case.

Acknowledgements

We acknowledge Dieter Waldshlaeger and Franz Badeck for giving us the opportunity to conduct this study. The Russian WWF supported the study morally and provided the GIS data for statutory reserves. We thank Alison Schlums for help with English editing of our paper.

References

Araujo, M.B., 1999. Distribution patterns of biodiversity and the design of a representative reserve network in Portugal. Diversity and Distributions 5, 151-163.

Ayensu, E., Van Claasen, D.R., Collins, M.,Dearing, A., 1999. International Ecosystem Assessment. Science 286, 685-686

Cowling, R.M.,Pressey, R.L., 2003. Introduction to systematic conservation planning in the Cape Floristic Region. Biological Conservation 112(1-2).

Cowling, R.M., Pressey, R.L., Lombard, A.T., Desmet, P.G.,Ellis, A.G., 1999. From representation to persistence: requirements for a sustainable system of conservation areas in the species-rich Mediterranean-climate desert of southern Africa. Diversity and Distributions 5, 51-57.

Cowling, R.M., Pressey, R.L., Rouget, M.,Lombard, A.T., 2003. A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa. Biological Conservation 112, 191-216.

Cowling, R.M., Pressey, R.L., Sims-Castley, R., le Roux, A., Baard, E., Burgers, C.J.,Palmer, G., 2003. The expert or the algorithm? - comparison of priority conservation areas in the Cape Floristic Region identified by park managers and reserve selection software. Biological Conservation 112(1-2), 147-167.

Ferrier, S., 2002. Mapping spatial pattern in biodiversity for regional conservation planning: Where to from here? Systematic Biology 51, 331-363.

Fonseca, G.A.B., Balmford, A., Bibby, C., Boitani, L., Corsi, F., Brooks, T., Gascon, C., Olivieri, S., Mittermeier, R.A., Burgess, N., Dinerstein, E., Olson, D., Hannah, L., Lovett, J., Moyer, D., Rahbek, C., Stuart, S.,Williams, P.H., 2000. ...following Africa's lead in setting priorities. Nature 405, 393-394.

Frazee, S.R., Cowling, R.M., Pressey, R.L., Turpie, J.K.,Lindenberg, N., 2003. Estimating the costs of conserving a biodiversity hotspot: a case-study of the Cape Floristic Region, South Africa. Biological Conservation 112(1-2), 275-290.

Gaston, K.J.,Rodrigues, A.S., 2003. Reserve selection in regions with poor biological data. Conservation Biology 17, 188-195.

Groves, C., Valutis, L., Vosick, D., Neely, B., Wheaton, K., Touval, J.,Runnels, B., 2000. Designing a Geography of Hope: A Practioner's Handbook for Ecoregional Conservation Planning. The Nature Conservancy, Arlington.

Hannah, L., Rakotosamimonana, B., Ganzhorn, J., Mittelmeier, R.A., Olivieri, S., Iyer, L., Rajaobelina, S., Hough, J., Andriamialisoa, F., Bowels, I.,Tiken, J., 1998. Participatory planning, scientific priorities, and land conservation in Madagaskar. Environmental Conservation 25, 30-36.

Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W.,Baltzer, M., 1998. Complementarity and the use of indicator groups for reserve selection in Uganda. Nature 394, 472-475.

Jaffre, T., Bouchet, P.,Veillon, J.M., 1998. Threatened plans of New Caledonia: Is the system of protected areas adequate? Biodiversity and Conservation 7, 109-135.

Lombard, A.T., Cowling, R.M., Pressey, R.L.,Mustart, P.J., 1997. Reserve selection in a species-rich and fragmented

landscape on the Agulhas Plain, South Africa. Conservation Biology 11(5), 1101-1116.

Mace, G.M., Balmford, A., Boitani, L., Cowlishaw, G., Dobson, A.P., Faith, D.P., Gaston, K.J., Humphreis, C.J., Vane-Wright, R.I.,Williams, P.H., 2000. It's time to work together and stop duplicating conservation efforts. Nature 405, 393.

Margules, C.R.,Pressey, R.L., 2000. Systematic conservation planning. Nature 405, 243-253.

Ministry of Environmental Protection and Nature Conservation of Russian Federation, 1995. The State report of environmental status of Russia in 1994, Moscow.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Fonseca, G.A.B.,Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Nikitina, E., 1995. Russia and other states of the Former Soviet Union: participation in multilateral environmental agreements. Intenational Institute for Applied System Analysis, Laxenburg.

Noss, R.F., Carroll, C., Vance-Borland, K.,Wuerthner, G., 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem. Conservation Biology 16(4), 895-908.

Pimm, S.L.,Lawton, J.H., 1998. Planning for biodiversity. Science 279, 2068-2069.

Pressey, R.L., 1994. Ad hoc reservations: forward or backward steps in developing representative reserve system? Conservation Biology 8, 662-668.

Pressey, R.L.,Cowling, R.M., 2001. Reserve selection algorithms and the real world. Conservation Biology 15(1), 275-277.

Pressey, R.L., Ferrier, S., Hager, T.C., Woods, C.A., Tully, S.L.,Weinmann, K.M., 1996. How well protected are the forests of north-eastern New South Wales? Analyses of forest environments in relation to formal protection measures, land tenure and vulnerability to clearing. Forest Ecology and Management 85, 311-333.

Pressey, R.L., Hager, T.C., Ryan, K.M., Schwarz, J., Wall, S., Ferrier, S.,Creaser, P.M., 2000. Using abiotic data for conservation assessments over extensive regions: quantitative methods applied across New South Wales, Australia. Biological Conservation 96, 55-82.

Pressey, R.L.,Logan, V.S., 1998. Size of selection units for future reserves and its influence on actual vs. targeted representation of features: a case study in western New South Wales. Biological Conservation 85, 305-319.

Pressey, R.L.,Nikholls, A.O., 1989. Efficiency in conservation evaluation: scoring versus iterative approaches. Biological Conservation 50, 199-218.

Red Data Book of Russian Federation / Plants, 1984. Rosselhozizdat, Moscow.

Red Data Book of the Russian Federation / Animals, 1983. Rosselhozizdat, Moscow.

Rodrigues, A.S., Cerdeira, J.O.,Gaston, K.J., 2000. Flexibility, efficiency, and accountability: adapting reserve selection algorithms to more complex conservation problems. Ecography 23, 565-574.

Rodriguez, L.O.,Young, K.R., 2000. Biological diversity of Peru: determining priority areas for conservation. Ambio 29, 327-329.

Scott, J.M., Davis, F.W., McGhie, R.G., Wright, R.G., Groves, C.,Estes, J., 2001. Nature reserves: Do they capture the full range of America's Biological Diversity? Ecological Applications 11, 999-1007.

Sierra, R., Campos, F.,Chamberlin, J., 2002. Assessing biodiversity conservation priorities: ecosystem risk and representativeness in continental Ecuador. Landscape and urban planning 59(2), 95-110.

Sokolov, V.E., Gorlenko, M.V.,Filonov, K.P. (Eds.), 1988. Itogi i perspektivy zapovednogo dela v SSSR. Academy of Sciences of the USSR, Moscow.

Tishkov, A., 2002. Biodiversity. In: Stolbovoi, V., McCallum, I. (Eds.), CD-ROM 'Land Resources of Russia'. International Institute for Applied Systems Analysis and the Russian Academy of Science., Laxenburg.

Venevskaia, I.,Venevsky , S., 2004. Hierarchical systematic conservation planning for a national level: example of Russia. 2. Setting conservation targets for Russian biodiversity hotspots. Biological Conservation, submitted.

Venevsky, S.,Venevskaia, I., 2004. Hierarchical systematic conservation planning for a national level: example of Russia. 1. Identifying national biodiversity hotspots using abiotic factors. Biological Conservation, submitted.

WWF,IUCN, 1994-1997. Centers of Plant Diversity: A Guide and Strategy for their Conservation. IUCN Publications, Cambridge**.**