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Monitoring wildlife richness — Finnish applications based on wildlife triangle censuses

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Studying human impact on ecological communities may require monitoring of wildlife communities on many spatial scales. Monitoring based on early and sensitive indicators of ecological change can be used administratively in ensuring the sustainability of populations. Wildlife richness is a multispecies concept which describes a general change in the abundance of an assemblage of wildlife species with reference to time and space. This is measured with the wildlife richness index (WRI), which is a sum of changes in the abundance of individual wildlife species. In this study, we present the administrative background of the needs for developing the multispecies monitoring. In order to meet the administrative need for ecological information, we introduce several applications, which are based on the interplay between the wildlife triangle scheme (WTS) and the concept of the WRI. In addition, we illustrate the usability of these applications with wildlife triangle data.

Introduction

The importance of sustainable harvesting of game animals has been strongly emphasised during the last decades. The monitoring of game populations is an essential prerequisite in ensuring this goal (*see* Sutherland 2001). As consequences of human activities vary in different spatial and temporal scales, methods and measures are needed to enable monitoring and analysis of game populations on several spatial scales (Wiens 1989). A good quality of monitoring information is a vital prerequisite for their applications.

Monitoring of game species has long traditions in Finland. The wildlife triangle scheme (henceforth WTS) was developed by the Finnish Game and Fisheries Research Institute in cooperation with the Hunters' Central Organization in 1988. It provides annual information on abundance levels and changes of some 30 wildlife species in about 1000 locations scattered throughout Finland (Lindén *et al*. 1996). The main goal of the WTS is to provide information on the populations to game administrators and local hunting associations as well as to individual game managers. In addition, the scheme produces versatile data for research purposes, for studies on e.g. population dynamics, predation, landscape ecology, and also for studies on biodiversity, at least concerning wildlife species.

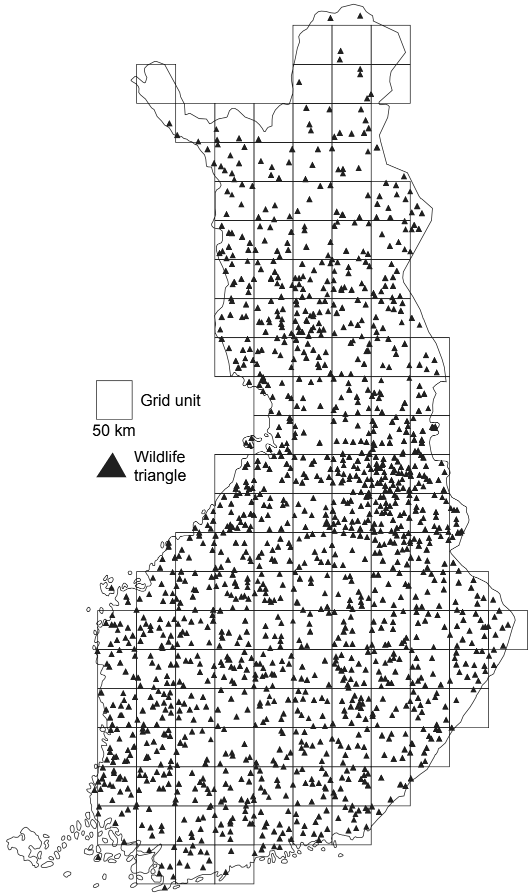

Fig. 1. The wildlife triangle census is the main method of monitoring many wildlife species in Finland. The coverage and density of triangles in forested areas is spatially and temporally representative.

To meet the administrative needs for monitoring sustainability in Finland, Lindén *et al.* (1999) proposed basic ideas for a tool to monitor wildlife richness, namely the wildlife richness index (henceforth WRI). It is a measure of the relative change in the richness of species assemblage, and it can be used for both spatial and temporal trend analysis in many scales. In addition, it can make use of available WTS-based data.

The concept of wildlife richness has some similarities with diversity. As for wildlife richness, two basic characteristics of diversity are species richness and species abundance. Various combinations of these characteristics have been introduced as diversity indices (Magurran 1988) and used as management tools. Another similarity is the idea of relating observed abundance to some other abundance. However, many diversity indices relate the abundance of individual species to the abundance of other species in order to characterise the community structure in relative terms. The WRI relates the abundance of each species to its abundance in reference time and space in order to describe the relative change in time and space among assemblages of species. Several diversity indices cannot be used in monitoring the change in abundance between assemblages. More flexible tools are needed, which enable the single- and multispecies trend analyses with reference to abundance, and if necessary, the weighting of species according to the study or management purpose. Lindén *et al.* (1999) outlined the desirable properties of a useful wildlife richness index and proposed basic ideas for the practical formulation of the index. However, they did not evaluate the properties of the wildlife richness index in respect to the management or research questions.

In this paper, various specifications of the WRI are made to meet and extend the ideas represented in Lindén *et al.* (1999): we present the administrative background of the needs for developing the multispecies monitoring. To meet the needs, we introduce several applications, which are based on the interplay between the WTS and the concept of the WRI. In addition, we illustrate the usability of these applications with wildlife triangle data.

Monitoring sustainable use

According to the general objectives of the Finnish Natural Resources Strategy (Ministry of Agriculture and Forestry 2001) game populations should be used sparingly and with careful consideration of their renewing and production potential. Another important principle is that hunting should not harm the diversity of game communities, i.e. the wellbeing of local wildlife in general should not be impoverished by hunting and game management. Sustainable harvesting of game animals should also be in balance with other forms of resource utilization and sources of livelihood.

To be aware of changes in game and wildlife communities, not only in individual game species, it was necessary to develop an indicator measuring multispecies wildlife richness. Thus there was a social order to develop an index describing wildlife richness (Lindén *et al*. 1999). The indicator is naturally an ecological measure, but it should serve administrative needs as well. This indicator should act as an "alarm bell" and unexpected changes in it should generate further investigations.

The wildlife triangle scheme in Finland

The WTS, founded in 1988, seemed to offer a basis for data to be used in monitoring wildlife richness, at least for forest wildlife species. In the following, we present the properties (and procedures), which enable the interplay between the WTS and the WRI in applications. On one hand, these procedures include the choice of species, which are informative in those aspects of wildlife richness, which are being studied. On the other hand, they include procedures, which are necessary to estimate animal abundances in various spatial and temporal scales.

Monitoring of many wildlife populations in Finnish forests is based mainly on the WTS (Fig. 1). The system provides information on 30 wildlife species, most of which are game species. It has a total of about 1600 permanent census locations (wildlife triangles); annually the censuses are carried out in 800–1000 triangles. During the last 15 years, approximately 50% of the triangles were censused at least 12 times. Almost 7000 voluntary assistants (mainly hunters) carry out the censuses (Helle *et al*. 1996).

The census line in the WTS forms an equilateral triangle with 4-km sides. The rigid shape and total length (12 km) of this line increase the probability that different forest habitats in the landscape are relatively well-represented. Although a linear transect has optimal sampling properties, triangular (and other closed) forms are more practical to the assistants to follow (Högmander & Penttinen 1996).

The choice of WRI species

Of the 30 species in the WTS, 17 were chosen to reflect wildlife richness (Table 1; Lindén *et al*. 1999). All of these species have forests as their primary habitat. There are many prey species, their predators and species which prefer different habitats and successional stages of forests. Many of these species are directly affected by hunting, and indirectly by species interactions or habitat alterations by, for instance, forestry and land use. In addition, the choice of species was restricted to those whose observations can be unambiguously interpreted and which are more or less regularly observed in the censuses (Lindén *et al*. 1999). Since most of the censused species are common game species and the censuses are largely made by hunters, misidentifications should be rare.

Although the number of species in a census is limited, the data may provide a good reflection of biodiversity describing the general welfare and richness of the wildlife in forests, since the species involved have different ecological roles in the assemblages. However, we acknowledge that there are many problems in using vertebrate species as indicators in ecology (*see* Landres *et al*. 1988). Even if the indicative role of some of

Table 1. The species, which are included into the WRI.

Included:
- mountain hare *Lepus timidus*
- red squirrel *Sciurus vulgaris*
- lynx *Lynx lynx*
- wolf *Canis lupus*
- wolverine *Gulo gulo*
- pine marten *Martes martes*
- red fox *Vulpes vulpes*
- stoat *Mustela erminea*
- white-tailed deer *Odocoileus virginianus*
- moose *Alces alces*
- wild forest reindeer *Rangifer tarandus fennicus*
- roe deer *Capreolus capreolus*
- capercaillie *Tetrao urogallus*
- black grouse *Tetrao tetrix*
- hazel grouse *Bonasa bonasia*
- willow grouse *Lagopus lagopus*
- otter *Lutra lutra*

Too randomly observed in winter (winter dormants):
- raccoon dog *Nyctereutes procyonoides*
- badger *Meles meles*
- brown bear *Ursus arctos*

Live in very restricted, and mainly water habitats:
- polecat *Mustela putorius*
- American mink *Mustela vison*

the species included in the WRI is relatively well known (Pakkala *et al*. 2003), further studies are needed to evaluate the value of these species in describing biodiversity in general. Until then, the results based on the WRI should be viewed as a general description of the state and dynamics of those wildlife and game species chosen to reflect wildlife richness.

The wildlife triangle census data

In the censuses, grouse are counted in the summer and mammal tracks in winter (Lindén *et al*. 1996). Grouse are censused in August using a three man chain flushing the birds from a 60-meter wide census belt. The results of the summer counts are converted into the estimate of grouse density (individuals/km^2) in forests. The efficiency of this count is about 70%–80% (Brittas & Karlbom 1990).

In winter, the tracks of mammals crossing the triangle line are counted, and an index of the abundances for each species is given as the track density (tracks/10 km/day). When grouse or track densities in the wildlife triangles are converted into abundance values of a larger area (e.g. an administrative unit, 50 × 50-km grid unit), an average of the track densities across the triangles in the unit is calculated for each species separately. This average value is called the abundance of a species in the unit.

Even if the results in the winter count do not depict animal densities, it is important to observe that when comparing the relative densities between two census units, we can use the *abundance ratio*: if we have 10 fox tracks/10 km/day in triangle A and 20 tracks/10 km/day in triangle B, the abundance ratio (A:B) is 0.50. We assume that there is no difference between the ratio of track densities and the ratio of animal densities. This is a relatively realistic assumption, especially if the units under comparison are near in time or space. If needed, there are many methods (e.g. mark–recapture methods) to test the plausibility of this assumption. Indices depicting wildlife richness are based solely on abundance ratios. For the details and accuracy of the monitoring method, *see* Lindén *et al*. (1996) and Högmander & Penttinen (1996).

The WR indices

The species-specific abundances in a unit are transformed into an index as follows: According to Lindén *et al*. (1999), the basic idea in constructing the WRI was to relate the observed abundance of each species to the abundance of the same species in another place, spatial scale or time point. The index value is based on the summation of these species-specific values. Within each unit, the species-specific value is obtained by calculating the ratio of abundances in a unit and in a reference area. Log-transformation is used to stabilise variation in the abundances. It is also useful for adjusting the sensitivity of the index. Zeros are dealt with by adding 1 to the ratio before the log-transformation. Finally, the species-specific values in the grid unit are added up. The resulting formula is:

$$\mathrm{WRI}_g = \sum_{i=1}^{S} \log\left(\frac{a_{it}}{R_{iT}} + 1\right) \qquad (1)$$

where WRI_g is the wildlife richness index in unit (e.g. grid) g, S is the number of species used, a_{it} is the abundance of species i in time point (or time period) t in unit g and R_{iT} is the abundance of the same species during time period T in the reference area. The choices of t and T are discussed below.

The WRI takes into account both species richness and abundance. In its basic form all species have equal weight in the index. Different weights can be set for species depending on the monitoring goals, and limits can be set for the maximum ratios of abundance in the grids and reference area. To prevent a single species from dominating the index, high species-specific ratios may be cut. Note that this introduces the idea of evenness into the index value. Lindén *et al*. (1999) limited the maximum value of the species-specific ratio to 3.0 (and therefore, the range of the ratio varies between 0 and 3.0). This means that the species-specific component of the index value can take e.g. values between 0 (as a result of $\log_2(0 + 1)$) and 2.0 (= $\log_2(3.0 + 1)$). The change in the ratio from value 1.0 (where the abundance of an individual species is the same in a unit and in its reference unit) to 3.0 (where a species is three times more abundant in a unit in respect to its reference unit) increases the species-specific component of the index value from 1.0 to 2.0 (using $\log_2$). The index component decreases by an equal amount, if the abundance ratio changes from value 1.0 to 0.

The WRI does not distinguish between species richness and abundance. For example, the same index value results in the two following situations: First, consider the case where there are only five species present, each being three times more abundant in the grid unit than in the reference unit. Second, there are ten species present, each equally abundant in the grid unit and the reference unit. In both cases the WRI value is the same (e.g. WRI = 10.0 as a result of ($5 \times \log_2(3.0 + 1)$ or ($10 \times \log_2(1.0 + 1)$)). Therefore, the values of the indices based on the summation of trends of many species tell only a little about the trends of individual species, a lesson pointed out by many authors (e.g. Link & Sauer 1996). Consequently, the values of the WRI should be interpreted together with the knowledge of the number of species and additional (e.g. species-specific) information.

Choosing the reference to meet different research needs

The flexibility of the WRI makes it a widely applicable tool in ecological research and management. The choice of the reference area and time point or period can be made to meet a specific need. Typical questions relate to (1) the temporal trend in the WRI, (2) the spatial trends in the WRI among areas, and (3) different combinations of spatial and temporal trends (i.e. the spatial trend of the temporal trend) in the short term. For graphical examples of these reference choices *see* Fig. 2. In addition, one may be interested in (4) the trend in the components of the WRI.

The temporal trend in the WRI

The monitoring of the temporal trend is needed for many kinds of management purposes. Generally, the WRI can be used as an indicator for trends possibly requiring management actions. A good resolution of the monitoring units is an important element in this kind of index use. Typically, the focus is on the monitoring of short term temporal change in specific area g in time point t in respect to some interesting time point (or period, *see* Lindén *et al.* 1999) T in the past.

If the temporal reference T in the calculations of WRI_g is fixed to a certain time point in the past, (e.g. the first value t_1 in the available time series), the role of the reference is simply to enable the calculation of the change in abundance for each time point (t_2, t_3, …, t_n) with respect to the fixed reference point or period (*see* Fig. 2a for a numerical example).

In addition to the fixed temporal reference point, moving temporal references for a_{it} (e.g. the average abundance of the two preceding time points ($R_{iT(t)} = (a_{it-1} + a_{it-2})/2$)) can be used (Fig. 2b). This kind of reference choice can be meaningful in cases where the recognition of rapid temporal changes in the WRI values is important. It may also be used in the smoothing of the variation in the time series of the WRI to clarify graphical analyses.

The similarity of temporal trends among areas in short term

In addition to monitoring temporal trends of wildlife richness within units separately, the WRI index allows monitoring of the differences between temporal progresses among units. This information can be used especially if interest lies in detecting the impact of a local factor in a unit g (e.g. a natural disturbance or management action) on the temporal trend in wildlife richness in respect to “control” units, which are assumed to be similar to the “test” unit in other ways.

A combination of moving temporal and spatial references can be used to describe these kind of characteristics in the WRI. For example, we may calculate wildlife richness values for any unit g in each time point t by using surrounding units in each time point as a reference ($t = T$). As a result, we get a time series of index values. A positive or negative trend in this time series reveals units where wildlife richness has developed differently than in the surrounding units (for example, *see* Fig. 2c). As abundance values reveal, the direction of the temporal progress in

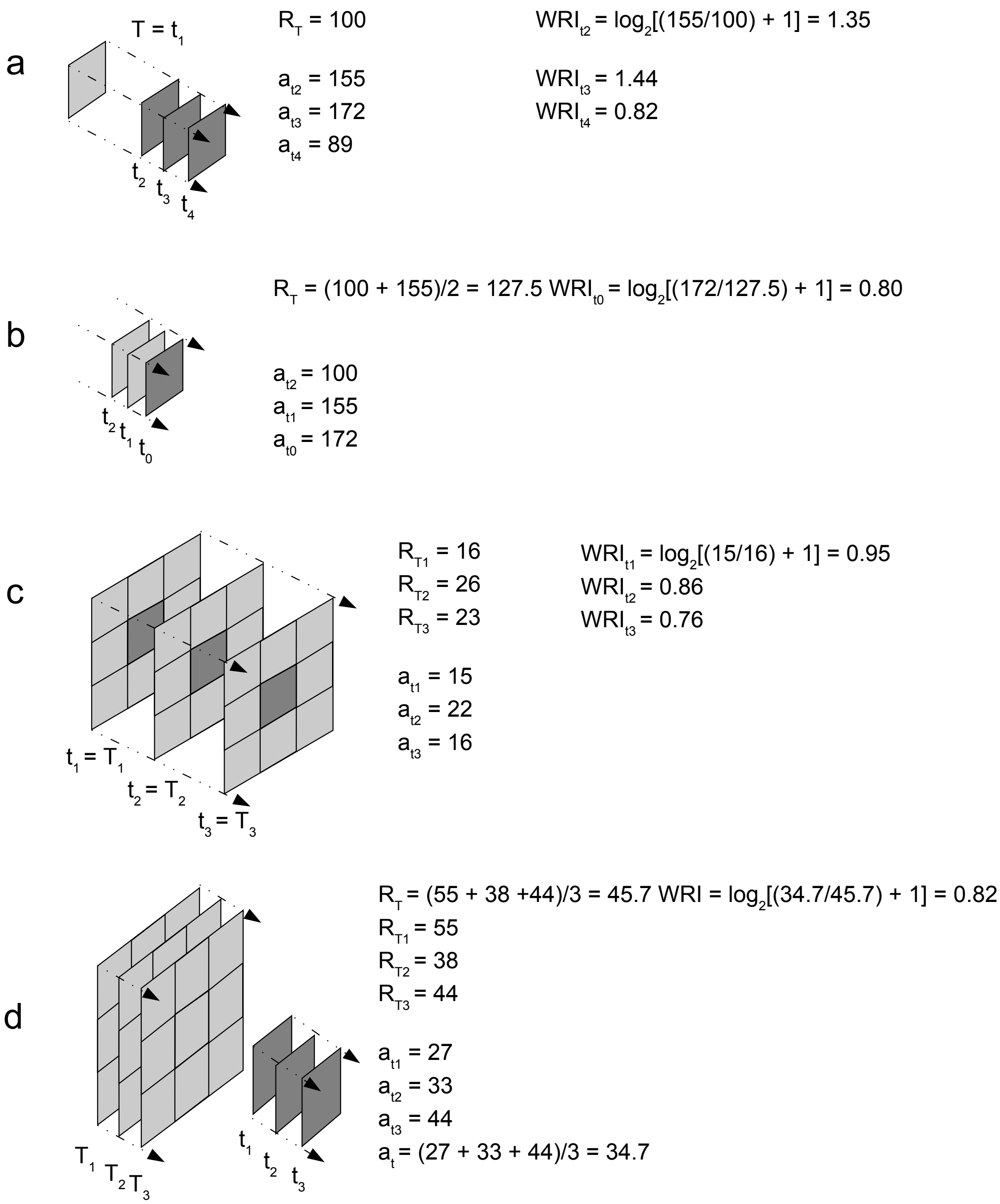

Fig. 2. Graphical illustration of the selected WRI reference choices with numerical examples. Each dark grey square unit denotes an area and time unit, for which the species-specific WRI component is calculated. A light grey unit denotes reference area and time unit. In **a**, the first abundance value in the time series (a_{t1}) is used as a fixed reference value (R_T) for the abundance values of the following years (a_{t2}, a_{t3}, a_{t4}), whereas in **b** the two previous abundance values (a_{t-1}, a_{t-2}) are used as a moving temporal reference (R_T). In **c**, the values in neighbouring units in the same time point ($t = T$) are used as reference. Finally, in **d** the spatial pattern of wildlife richness is calculated for each (but here only one) unit during a long time period. In **c** and **d**, the reference values (R_T) are averages of eight and nine grid abundances, respectively.

consecutive values is similar in both units, even though the abundances are on a different level especially in t_2 and t_3. This progress in the difference between abundance levels is emphasised in the time series of the index values.

The spatial trends in the WRI

The spatial pattern of wildlife richness provides an interesting viewpoint on many kinds of research and management problems. Variation in the spatial pattern of wildlife richness in any time point may result from the combined effects of several factors, e.g. multi-species synchrony in population dynamics, weather conditions or other disturbances. However, little is known about these factors and great care must be taken in the analysis of spatial patterns, especially in short term.

The variation in the long-term spatial pattern of wildlife richness may reflect more strongly the general impact of the regions' natural characteristics (soil, vegetation, weather) and other relatively unchangeable factors on wildlife. It may also reveal long-term effects of anthropogenic activities affecting landscape characteristics and wildlife. The WRI can be used to describe the general state of species assemblages in studies, where these hypothetical associations are being tested.

The reference in the calculation of spatial WRI can be varied according to the study objectives: if the purpose is to generally describe the spatial pattern of WRI_g, the same fixed temporal and spatial reference R_{iT} for every study unit g enables comparisons between every pair of units (for a numerical example, *see* Fig. 2d). Note that index values are calculated separately for each species. Therefore, even though the sum (or even the species-specific values) of the observations among a group of species may be the same in the grids, but values relate to different species (= species-specific rank of abundance is reversed), the relative abundances in respect of the reference are different, and so are the WRI values. This kind of difference (which can occur in time or space) cannot be detected using those diversity indices (e.g. the Shannon index), which only describe the structure of community and refer species abundances to other species in the same unit. If the purpose of the monitoring is to recognise rapid changes (edges) in the spatial pattern of WRI_g values, surrounding units (e.g. four or eight units in a square grid) as moving reference R_{iT} will emphasise more strongly the local variation.

The trends of the WRI components

Species and species groups are known to have different associations with disturbances (e.g. anthropogenic activities) and they may have different variations in time or space. Therefore, actions may have positive effects on species and some species groups, and negative effects on others. In order to describe the trends in the components of the WRI value, index values can be calculated separately for different sets of species assemblages (i.e. species groups) by summing species-specific WRI_g values. Then, these indices can be compared with each other. However, if needed, an index value can be calculated by using species or sets as references.

Which shape and resolution for the units?

In addition to the compatibility of the reference and the choice of species in respect to different research and management needs, one has to decide upon the shape and the resolution of the study units. The choice of a scale is important in several contexts (e.g. Frankham & Brook 2004).

A basic spatial unit in this (and many other) monitoring program is based on a grid. There are many practical reasons for this choice. First, every unit has the same area, which makes it easier to compare unit-specific numbers. Second, point data (i.e. WTS data) in the grid units can be relatively easily converted to values in larger scale. In addition, it is computationally efficient to make calculations and combine grid-based

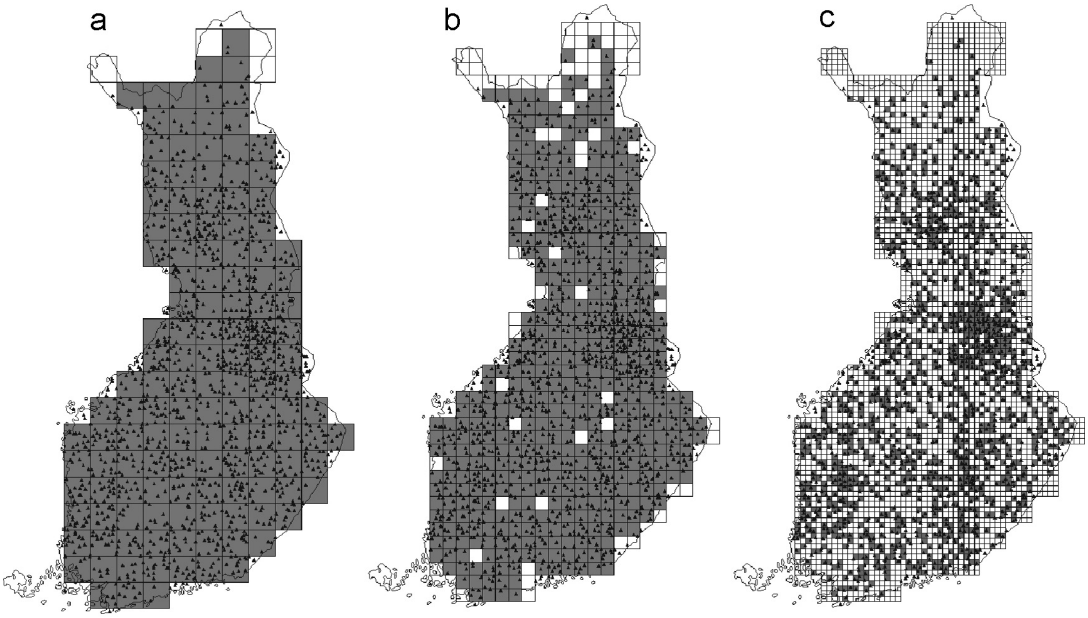

Fig. 3. The coverage of wildlife triangle data in respect of the size of the grid units. In **a** the size of a grid unit is 50 × 50 km, in **b** 25 × 25 km, and in **c** 10 × 10 km. Dark squares indicate grid units with at least one census location.

data sets from other monitoring systems (e.g. satellite images). However, the shape of the area units can be modified if needed. For example, the purpose of the study may be to study the effect of human activities on wildlife, and these activities are known to differ between administrative units. In addition, the data describing human activities may be available only from these area units. In this case, a WRI estimate can be calculated according to administrative units.

The choice of the spatial and temporal resolutions of the unit is ideally made according to study objectives. The principles of the WRI can be applied to the objectives concerning richness of wildlife in the local census locations (e.g. wildlife triangle in the WTS) during the season, but the WRI can be a useful concept at the regional level and across years, as well.

The availability of data (sample size) and the required precision for the results may constrain the ideal choice of the spatial and temporal resolutions in practise (for example, *see* Fig. 3). In other words, the biases of the estimates tend to be higher and the precision lower, when sample sizes are small. These aspects need to be considered in every situation separately.

Applications of the WRI in Finland

In the following we illustrate the different aspects of the applicability of the WRI in different contexts using wildlife triangle data from 1989–2003 with 50 × 50-km grid resolution. Note that this time series is more than twice as long as known cycle lengths in population dynamics (Lindén 1988). Therefore, even though our only purpose here is to use statistical tests to describe the properties of the data, the observed significant trends in the index values may not be entirely explained with natural variation.

Procedures

We used species-specific mean abundances for every year in 1989–2003 and the mean abundances of the same grid unit over the years 1989–1994 as a reference time period in our calculations of the temporal trend in the index for each grid unit and year.

To illustrate the spatial distribution of the WRI in Finland, a long-term (13 years) average abundance was calculated for each species in the units. Species-specific average abundances in the country were used as reference values.

A similarity between the WRI in a unit and its neighbours at the same time was calculated by using each unit as an independent observation and the surrounding eight neighbouring units as a reference area (in the bordering units, the number of reference units depended on the availability of units). An index value was calculated for every year and unit separately. As a result, a time series of index values was created, enabling trend analysis (for methods, *see* e.g. Thomas 1996).

We calculated index values separately for ungulates and large predators using the same data as in the calculations of the spatial trend of the WRI. The ratio of ungulates index to large predators index reflects the relative richness between predators and their prey, even though a low number of large predators (and sample sizes in northern Finland) makes abundance estimates relatively unreliable.

To estimate the grid-specific index values with respect to the variation in the wildlife triangle (point) data, we used non-parametric percentile bootstrapping (e.g. Manly 1997). In this procedure data points were re-sampled 1000 times in each grid unit with sample size (n). The index value was calculated each time to estimate the expected value and coefficient of variation (CV) of the sampling distribution for the index. The purpose of this procedure was only to demonstrate the sensitivity of estimates to the variation in the data.

The average number of wildlife triangles in each year is about five triangles per 50 × 50-km grid. The sample size in northern Finland, however, is often very small. This may especially affect detectability or the bias and precision of abundance estimates of rare species. We assumed that a similar detectability (e.g. due to moving behaviour) holds on both spatially and temporally. Under these assumptions, we present the following results:

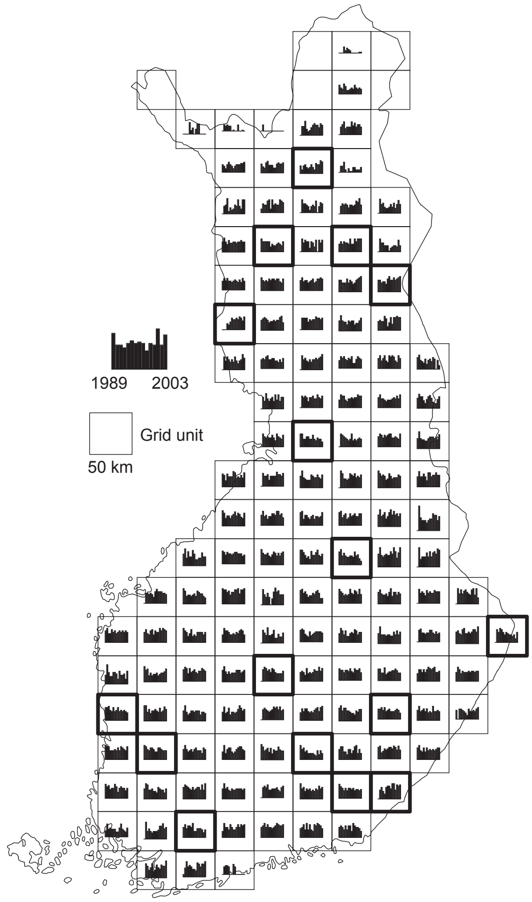

Fig. 4. Temporal wildlife richness for each year in grids in 1989–2003. The maximum index value among the grid units was 6.8. The grids with statistically significant trends are shown in bold borders.

The WRI in a specific area in the short term in Finland

The Spearman's rank correlation coefficient (as well as the slope of the linear regression trend line) of the WRI values in 133 grid units was significantly ($p < 0.05$) negative in 11 units and positive in five units. The expected number of locally present species in the wildlife triangles varied from 4 to 10 (out of the 17) and remained relatively stable over time. Therefore, variations in wildlife richness in Finland in past years seemed to depend more on changes in the abundance of species than in the number of species.

However, species numbers have also decreased in almost all of the grids with significant negative trends. In most of these cases, lynx (*Lynx lynx*), stoat (*Mustela erminea)*, pine marten (*Martes martes*) or red squirrel (*Sciurus vulgaris*) tracks have not been detected during

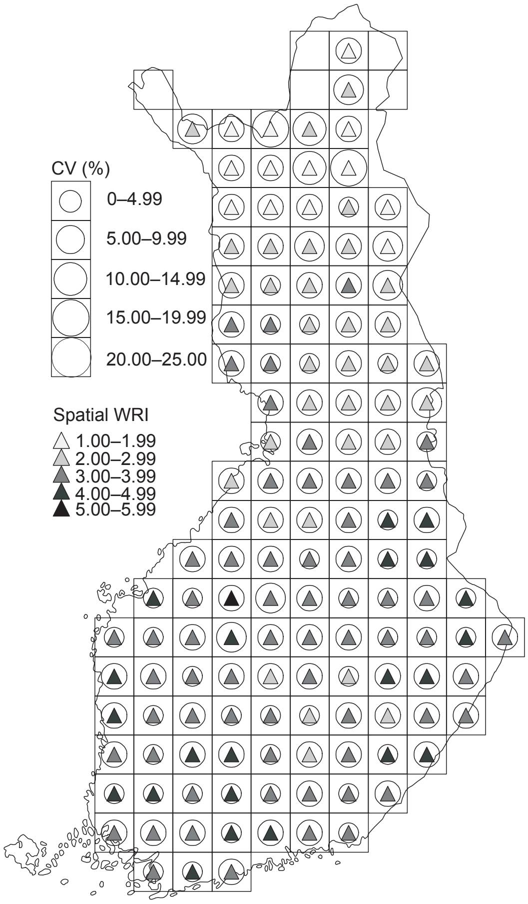

Fig. 5. Spatial wildlife richness in Finland 1989–2001. The darker the triangle, the greater the index value in the grid unit. The size of the circle describes the coefficient of variation (CV) of the estimate based on the bootstrapping of the data.

the censuses in recent years (Fig. 4). It is important to further study the factors that explain these trends. The changes in forest habitats and other anthropogenic changes from a local to a regional level are plausible explanations for these changes.

The general spatial trends in the WRI among areas in long term in Finland

According to the data (Fig. 5), regional wildlife richness was greatest in eastern and north-eastern Finland. There were also some rich grids in southern Finland, for example in the province of South Häme. The pattern of the regional WRI was very predictable to a wildlife expert. The rich grids in eastern Finland lie in the mid-boreal forest vegetation zone (Ahti *et al*. 1968), which also seems to be the transition zone between southern and northern fauna. The percentage of forests is high in the east, and decreases westwards where agricultural land, roads and settlements are more common. One explanation for the higher wildlife richness in eastern Finland is that it is near to Russian Karelia, where forest landscape is more natural, and where both species richness and abundance are higher than in Finland. The belt of rich grids which begins in eastern Finland along the Suomenselkä may be explained by the connections to the taiga fauna (Lindén *et al*. 2000).

The similarity of the temporal trend among areas in the short term in Finland

Linear trend slopes of the time series of these index values were in most units relatively similar to the trend slopes of the WRI values in a specific area in the short term. Statistically significant ($p < 0.05$) positive Spearman's correlation coefficients of the index time series were found in 11 units and negative ones in 9 units (Fig. 6). Four of the locations with significant negative trends and four with positive trends had also significant trends regarding the WRI values in a specific area in the short term.

Spearman's correlation coefficient of the similarity was significant in 12 units, where the WRI values in a specific area in the short term did not have a significant trend (Fig. 6). The analysis revealed five units where the correlation coefficient was significantly negative. This indicates that in these units the slope of the temporal trend in wildlife richness was consistently more moderate than in the surrounding units, even if its temporal trend in respect to its own richness level in the past was not significant. Contrary to that, there were seven units where the trend slope was consistently steeper. In addition, there were eight units which had a significantly consistent trend in time in respect to its own past, but which did not have different trends than the surrounding units.

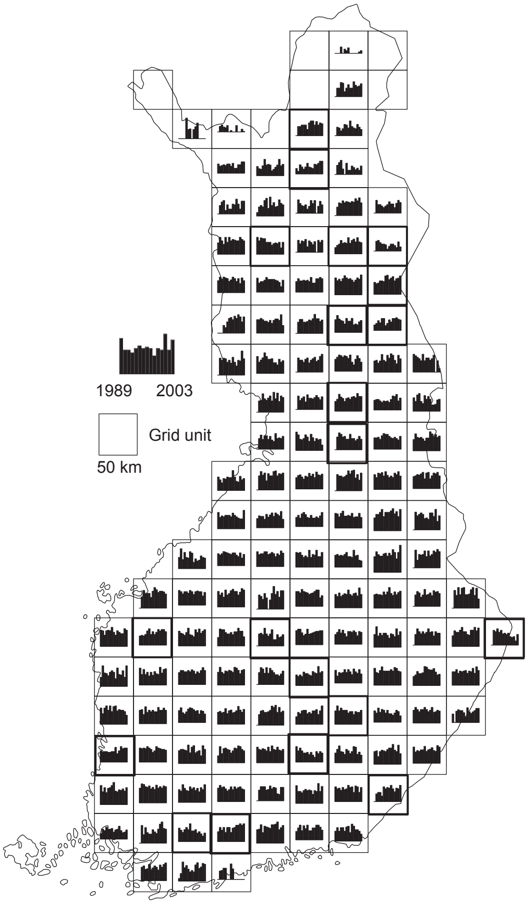

Fig. 6. Similarity of the temporal trend among neighbouring grids in 1989–2003. The statistically significant trends are shown in bold borders.

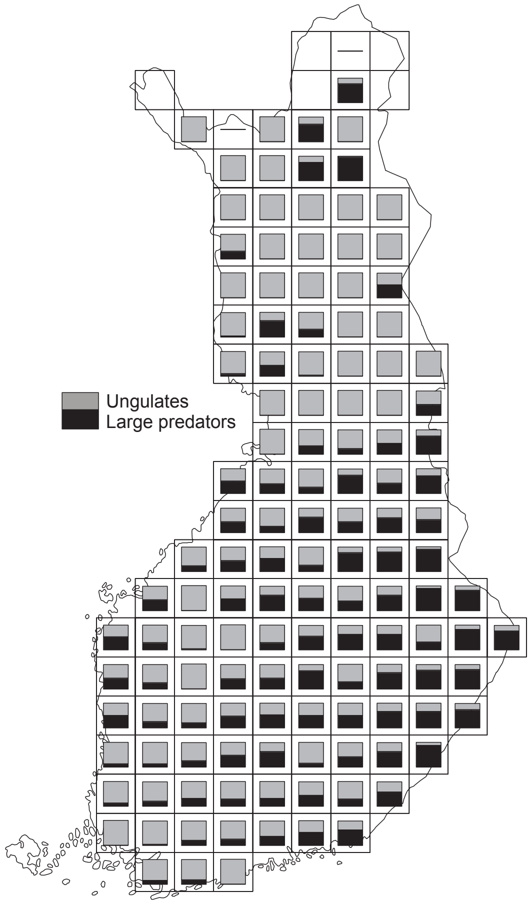

Fig. 7. The relative richness between large predators (wolf, wolverine and lynx) and their prey ungulates (moose, wild forest reindeer, white-tailed deer and roe deer) in the spatial richness index in 1989–2003.

The relative richness between large predators and ungulates

The relative richness of large predators in respect to ungulates (prey) changes gradually in an east–west direction (Fig. 7). An interesting "edge" in the relative proportions can be seen between central and northern Finland near the administrative border of the reindeer husbandry.

Conclusions

From the management point of view, it is important that the index is sensitive to the variation in the chosen aspect of wildlife richness, enabling location of any possible alarming trends in time or space. This is a challenge for any monitoring tool in wildlife management. We believe that the interplay between wildlife triangle data and the WRI based on meaningful choice of references can satisfy both of these criteria, and that the applications can be successfully utilised as a tool in Finnish wildlife management. As these examples point out, the properties of the WRI, including the sensitivity and specificity, can be varied in a flexible way to meet administrative needs for ecological information.

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