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How much compensation do we need? Replacement ratio estimates for Swiss dry grassland biotopes

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ABSTRACT

Development pressure on reserve networks in densely populated countries may lead to the decision to allow for replacement compensation. Replacement ratios used for specifying replacement compensation are usually based on expert judgment. In contrast, we propose a method to estimate replacement ratios based on the set covering framework. The method is applied to presence–absence data of vascular plants of the dry grassland inventory of Switzerland. For the replacement of 60% of a patch's high conservation value species by the same vegetation type (''in-kind'' compensation), the estimated replacement ratios are <5 for most vegetation types. These ratios are comparable with replacement ratios usually used in practice. Our replacement ratio estimates for replacement by another vegetation type (''out-of-kind'' compensation) are considerable higher than proposed by the literature. For oligotroph dry grassland associations, the replacement rations are extremely high, so that these associations have to be considered irreplaceable. The estimated replacement ratios provide a good starting point for designing compensation measures for unavoidable losses in a reserve system. However, additional biodiversity conservation goals should be considered when designing replacement compensation in practice.

1. Introduction

Biodiversity conservation and especially offsetting the current worldwide biodiversity loss is one of the biggest challenges pointed out by the ''Convention of Rio'' (UNEP, 1992). Patch-based reserve networks provide an important approach for facing this task (Margules and Pressey, 2000; Margules and Sarkar, 2007; Noon et al., 2009; Pressey et al., 2007). The selection of patches for such networks is based on principles of hotspot (Orme et al., 2005), complementarity and representativity (Kati et al., 2004). In densely populated countries, economic pressure may lead to the decision to release patches from an established reserve network and compensate for the loss with the inclusion of other patches (Rundcrantz and Skärbäck, 2003). Furthermore, patches sometimes are excluded from an existing reserve network because they have been degraded by disturbance (Drechsler et al., 2009) or by climate change (Araújo et al., 2004).

Several countries have issued guidelines on how to deal with threats to protected patches in a reserve network (Review by Rundcrantz and Skärbäck (2003) and Kägi et al. (2002)). All strategies follow the same principles: If possible, the threat should be removed. If this is not possible, the damage should be mitigated, and if this is not possible, ''replacement compensation'' is required. Replacement compensation is defined as ''environmental compensation for lost environmental values implemented in another functional context (off-site and/or out-of-kind compensation)'' (Rundcrantz and Skärbäck, 2003).

In practice, reserve network planners are rather often faced with the task of replacement compensation. Replacement ratios are a simple and popular approach to determine how to compensate (Allen and Feddema, 1996; Cabeza and Moilanen, 2006). The replacement ratio for two vegetation types is the number of patches of one vegetation type needed to replace a given fraction of the species of a patch of the other vegetation type. Replacement ratios can be used in combination with further objectives, e.g. optimising spatial configuration (Miller et al., 2009), and assumptions about metapopulation dynamic can be integrated in replacement compensation (Cabeza, 2003; Cabeza and Moilanen, 2003; Nicholson et al., 2006).

Expert judgement and negotiations among stakeholders provide pragmatic approaches to determine replacement ratios (Allen and Feddema, 1996; Calörtscher, 1996; Righetti, 2002). Since the results of these approaches are usually vague, other approaches have been proposed. Replacement ratios based on a valuation of a number of ecological factors has been suggested by Henz (1998) and Schnapauff (1998). A method to estimate compensation payments was proposed by Schemel et al. (1995). Replacement compensation was quantified based on measurements of

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ecological services by Brinson and Rheinhardt (1996) and by Roach and Wade (2006). Even though replacement compensation is about replacing environmental value, there is hardly any research about estimating replacement ratios for biodiversity networks directly from ecological data.

Our research addresses this lack of research and presents a method for estimating replacement ratios directly from ecological data. In order to do so, several basic assumptions of the replacement compensation approach are accepted, i.e. that patches in a reserve network can be replaced, that patches can be created, and that the new network with replaced patches has a predictable future. These assumptions are debated in the literature and therefore warrant some discussion.

Replacement compensation accepts the idea, that a high-quality patch may be replaced by a larger patch of lower quality. This idea has been criticised by conservation biologists (Shafer, 1990). The position pursued in this paper is neither to advocate for this idea nor to refuse it.

In practice, all valuable patches are usually already protected and therefore not available for compensation. Therefore, replacement patches must be created by restoring mediocre patches. Unfortunately, this restoration normally happens after the patch has been integrated into the reserve network. To counteract the risk that the restoration is not successful, a "rate of interest" can be set to compensate for regeneration time and risk. Furthermore, biotope types that have an extremely long regeneration time – as some dry grassland types – can be excluded from replacement compensation (Schemel et al., 1995). An alternative is to compensate in advance by the means of mitigation banking (Bendor, 2009; Hartig and Drechsler, 2009; Wagner, 2006).

The long-time success of replacement strategies and projects has been evaluated from various perspectives. The application of the US Clean Water Act has been evaluated in regard to the area per wetland type (Allen and Feddema, 1996; Harper and Quigley, 2005) and in regard to the "no net loss principle" (Harper and Quigley, 2005). From the theoretical side, the use of simple "currencies" that facilitate trading between biodiversity and development was criticised fundamentally (Walker et al., 2009), and the use of numerical balancing methods was characterised as "pseudo-scientific hocus-pocus" (Böhme, 2005). Long-time success of replacement strategies respective to measures based on irreplaceability (Ferrier et al., 2000), probability of survival (Drechsler, 2005), efficiency and retention (Pressey et al., 2004; Cabeza and Moilanen, 2006) has been investigated by simulation studies (Meir et al., 2004; Pressey et al., 2004) or by stochastic dynamic programming (Drechsler, 2005). Models might also include assumptions about future climate change or degradation probability (Snyder et al., 2005; Moilanen et al., 2009; Turner and Wilcove, 2006). The evaluation studies demonstrate that a reasonable compensation strategy does not guarantee the maintenance of biodiversity targets through time, and that learning from monitoring of compensation projects is important.

The selection of the surrogates used to measure biodiversity has a strong influence on replacement ratios. Surrogates can be measured by focal species (Noon et al., 2009), vegetation or habitat types (Cuperus et al., 1999; Righetti, 2002), ecological functions (Brinson and Rheinhardt, 1996) or vascular plants (Sætersdal et al., 2004). Biodiversity surrogates may be weighted according to their phylogenetic relationship (Rodrigues and Gaston, 2002), irreplaceability and vulnerability (Lawler et al., 2003). A surrogate always represents only one aspect of biodiversity and is therefore a simplification (Bonn and Gaston, 2005). In the dry grassland data set, biodiversity surrogates are vascular plants and vegetation types.

Ideally, stakeholders should be able to participate in the selection process of reserve patches (Cowling et al., 2003; Schenk et al., 2007) and replacement compensation. Indeed, the success of a biodiversity conservation program, i.e. a patch-based reserve system, is greatly affected by the information and education provided to the decision makers (Luz, 2000) and by an appropriate organisation of participation (Review by Reed (2008)). The method proposed in this paper provides estimates for replacement ratios which can serve as a starting point for replacement compensation. There is much room for stakeholder participation in the subsequent process of designing compensation measures based on these replacement ratios and additional biodiversity conservation goals.

2. Material and methods

2.1. Study area and vegetation data

The dry grassland inventory of Switzerland is the basis of the biodiversity conservation project for dry grasslands, realised by the Federal Office for the Environment (FOEN). The goal of this project is to secure a network of dry grassland patches through management contracts between government and farmers. The inventory has gained official status in February 2010 (Schweizerischer Bundesrat, 2010). The inventory describes 13,531 dry grassland patches varying in size from 488 m^2 to 1.1 km^2 (mean: 1.8 ha, median: 0.9 ha), adding up to 250 km^2 (0.6% of Switzerland) (Fig. 1). The patches were identified and delineated based on aerial photos and fieldwork. Within each patch, a circular sampling plot of 3 m radius ($\sim$30 m^2) was placed to best represent the dominant vegetation type. Fieldworkers then recorded the presence–absence data for typical dry grassland species within each plot. The methodology is described in detail in Eggenberg et al. (2001). Fieldwork was conducted between 1995 and 2006. Additional information about the project is available on the Internet site *www.bafu.admin.ch/tww*. Data is stored at FOEN and at the Datacenter Nature and Landscape (*www.wsl.ch: Search term "DNL"*).

Eighteen vegetation types and 466 dry grassland species have been identified. The characteristics of the vegetation types are summarised in Appendix A. The definition of the vegetation types primarily follows Zoller (1954) and Ellenberg (1996) and the CORINE project (Commission of the European Communities, 1991). Because of conservation considerations, the two additional vegetation types *ll* (low diversity grasslands of low altitude) and *lh* (low diversity grasslands of high altitude), were identified and the *Mesobromion* sensu Ellenberg (1996) was subdivided into the four vegetation types *mbae*, *mb*, *mbxb* and *mbsp*. Experts assigned a high conservation value to 13 vegetation types (column "Value", Appendix A).

Of the 466 dry grassland species, experts classified 358 species as high-value species and 108 species as low-value species. Low-value species are especially the tall forbs, the ruderal plants and the indicator species for *Arrhenatheretalia, Nardion* and dry fringes (*Origanetalia*).

Even though the data has been collected for practical conservation purposes, the data set is well suited for investigating our research questions. The sampling plot size of 30 m^2 is reasonable for dry grasslands (U. Graf, WSL, personal communication) and lies within the 10–100 m^2 range proposed by Zoller (1954). Because the sampling was part of a biodiversity conservation project, the vegetation types of high conservation value are highly differentiated. Therefore, the results for these vegetation types are more detailed. The data set does not statistically represent the dry grassland area with its species since the sampling method has been designed to represent the dry grassland vegetation types within Switzerland. However, our study focuses on vegetation types, so that this is not an issue.

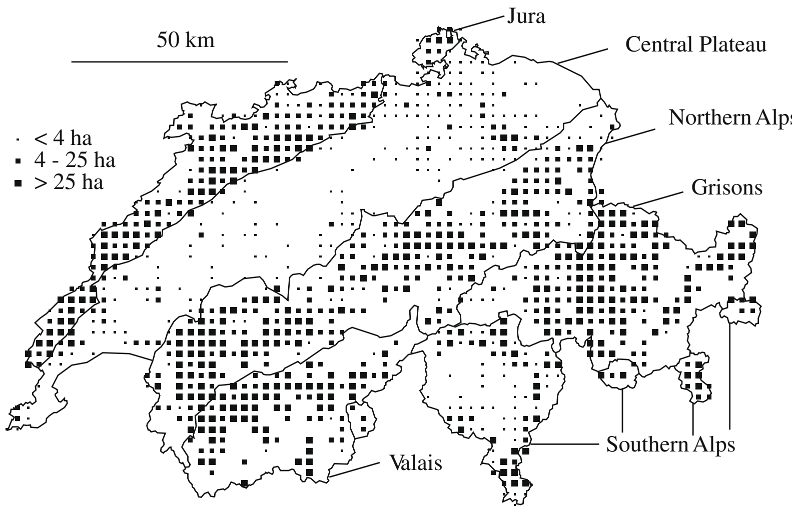

Fig. 1. The case study area is Switzerland. Black squares represent the amount of dry grassland area per 5 km × 5 km grid cell. Biogeographical regions (Gonseth et al., 2001) are indicated.

2.2. Calculation of replacement ratios

The replacement ratio for a pair of vegetation types refers to the number of plots of a given vegetation type R needed to replace one plot of a given vegetation type L so that the replacement fraction reaches value T. The replacement fraction of a replacement action (the replacement of one lost sampling plot with one or more replacement plots) is the fraction of species of interest (respective to a given reference list) of the lost plot present in at least one of the replacement plots.

The replacement ratios and the corresponding uncertainty ranges are estimated by randomly drawing plot sets of different size and comparing their species composition with a randomly drawn reference plot. In other words: We solve in a statistical way the "partial set covering problem", i.e. the search for a minimum number of plots so that at least a given fraction of a given species list is present in at least one plot (Gandhi et al., 2004). The algorithm is described in detail in Table 1.

In order to demonstrate the algorithm, it was applied to the replacement of one plot of high-nutrient semi-dry grassland (*mbae*, Appendix A) by 1–10 plots of true semi-dry grassland (*mb*, Appendix A). The results are summarised in a box-and-whisker diagram (Fig. 2). Both axes were log-transformed to map power functions as lines (Arrhenius, 1923; Dengler, 2009). A linear regression has been calculated to analyse the change in the species replacement fraction with increasing number of replacement plots. For a target species replacement fraction of 60%, for example, 2.6 plots of true semi-dry grassland (*mb*) are required to replace one plot of high-nutrient semi-dry grassland (*mbae*). With a 50%-uncertainty range, the value lies between 1.6 and 4.2 plots.

For the display of the replacement ratios in the results, the vegetation types are arranged according to their similarity. To determine the similarity, a hierarchical cluster analysis based on replacement rations was performed in which vegetation types with a low average replacement ratios are grouped closely together (e.g., *mbxb* replaces *xb* with a replacement ratio of 3.70. The reverse replacement has a ratio of 3.84, and the average is 3.77). Complete linkage is used for the cluster analysis; i.e., the distance between two clusters is defined as the lowest replacement ratio between

Table 1
Algorithm for calculation of the replacement ratio for a pair of vegetation types.

Input	L is the vegetation type of a lost plot, while R is the vegetation type of a replacement plot. A matrix is produced with all plots of the dry grassland inventory in columns, all species of a reference species list in rows and presence–absence data in entries. T is the target species replacement percentage, i.e. the fraction of species of the lost plot that should be present in at least one replacement plot
Algorithm	
Step 1:	From the set of plots, $n_p = 10$ samples of size $n_q = 100$ are taken. A sample item s_{pq} (with $p = 1, \ldots, n_p$ and $q = 1, \ldots, n_q$) is composed of one plot belonging to vegetation type L, and p plots belonging to vegetation type R. All samples are taken with replacement
Step 2:	The fraction of species of the lost plot that occurs in at least one replacement plot is denoted as c_{pq}. For each sample item s_{pq}, the species replacement fraction c_{pq} is computed as $c_{pq} = n_{LR}/n_L$, where n_{LR} is the number of species of the reference set that occurs in the lost plot and in at least one replacement plot, and n_L is the total number of species of the reference species list that is present in the lost plot
Step 3:	Intersection a and slope b of the linear model ${}^{10}log\, c_{pq} = a + b * {}^{10}log\, p + \varepsilon_{pq}$ are computed with the $n_p * n_q$ sample items s_{pq} and ε_{pq} as error term
Step 4:	Solve equation ${}^{10}log\, T = a + b * {}^{10}log\, p$ for p, where T is the target species replacement fraction, and a and b are taken from step 3. The result is the required replacement ratio $p = 10^{(T' - a)/b}$ with $T' = {}^{10}log\, T$
Step 5:	Calculation of the uncertainty range of the replacement ratio The 50%-uncertainty range should contain the half ($u = 0.5$) of the sample items around the median. It is assumed that residuals follow normal distribution (a) Calculation of standard deviation s of residuals of the sample items (b) Calculation of $q(u)$, the multiplicand for s. It is the difference between the $(0.5 + u/2)$-quantile and the median of the normal distribution In R-language (R Development Core Team, 2009) $q(u)$ is $qnorm(0.5 + u/2)$. E.g. $q(0.5) = 0.674$ (c) The lower limit of the uncertainty range is $p/\Delta x$, and the upper limit is $p * \Delta x$ with $\Delta x = 10^{q(u)*s/b}$, where b is the slope from step 3

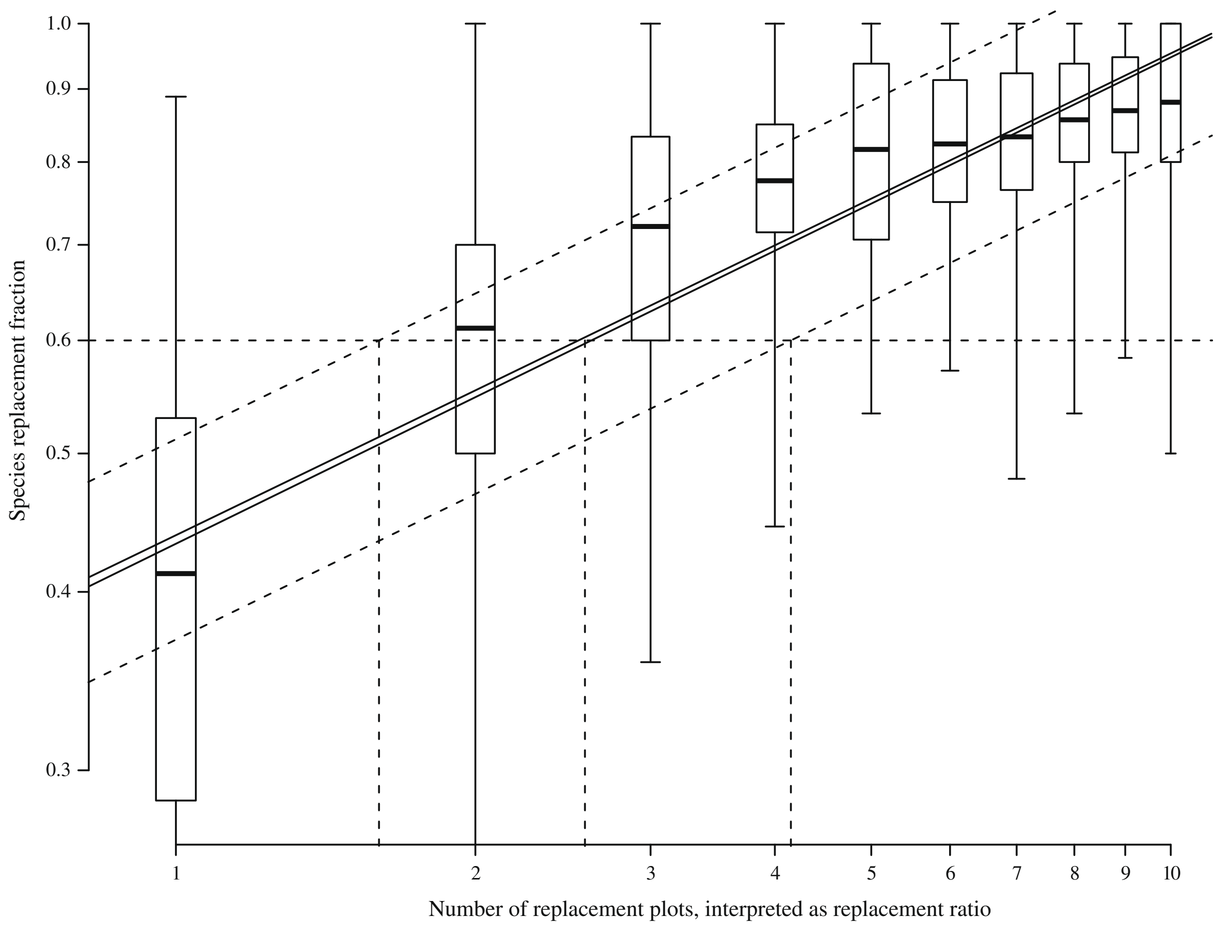

Fig. 2. Illustration of the replacement ratio calculation for the replacement of a high-nutrition *Mesobromion* (*mbae*) by a true *Mesobromion* (*mb*) with replacement fraction 60%. The box-and-whisker plots summarise the number of replacement plots by 0.75- and 0.25-quantiles (top and bottom of boxes), median (bold lines), maximum and minimum (whiskers). The double line shows the linear regression line according to the power function (i.e. both axis are log-scaled). Dashed lines parallel to the regression line show the 50% prediction interval. Horizontal and vertical dashed lines show the replacement ratio (2.6), and the lower (1.6) und upper (4.2) limit of the 50%-uncertainty range.

all pairs formed by one vegetation type of each cluster (Sneath and Sokal, 1973).

Two principles must be accepted so that plot-based replacement ratios can be used as area replacement ratios. First, the conservation goal should demand that at least a given fraction of the lost biodiversity surrogates (e.g. species) must be present in the replacement patches (retention principle). Second, the patch can be considered as a group of many equal-sized small plots, and each plot will be replaced individually (additivity principle). However, the ecological implications of this second principle remain somewhat unclear and we have not been able to investigate them with the dry grassland data set.

For the presentation of the results we choose a replacement fraction of 60%. Preliminary clarifications with decision makers showed that this choice seems reasonable for the national dry grassland conservation program, but higher or lower fractions might also warrant some discussion.

2.3. Estimating the effects of four variables on replacement ratios

We analyze four variables in detail in order to better understand their effect on the replacement ratio estimation:

(1) The effect of the species set (representing the biodiversity surrogate) on the replacement ratios is analysed by comparing the results for two species lists: The results for all 466 dry grassland species and the results for the 358 species of high conservation value. The effect is especially clear if replacement ratios are classified into four groups. These groups are defined by vegetation types of high/low value as lost/replacement plot. Based on theoretical considerations (see Appendix B for more detail) the following assumption holds true: If a vegetation type of low value is replaced by a vegetation type of high value, the lost plot has more species of low conservation value than the replacement plot. In this case, the replacement ratio calculated with high-value species is lower then the replacement ratio calculated with all dry grassland species.

(2) The effect of the replacement fraction on replacement ratios is shown in Fig. 2 and is a consequence of the formula in Table 1, step 4. Replacement ratios increase linearly with increasing replacement fraction.

(3) Spatial autocorrelation might be an issue with the data set of the Swiss dry grassland inventory. Therefore, spatial autocorrelation was analysed with two approaches. First, we calculated the linear regression between distance (in km) and species replacement fraction of plot pairs composed of one lost plot and one replacement plot. Analysis was based on random samples of 100 plot pairs. A plot was used only once to insure independence. The degree of autocorrelation is expressed by the slope of the regression line, measured as a decrease or increase in the species

replacement fraction per 100 km. Two-sided *t*-statistics were used to test for autocorrelation (slope unequal zero). Second, we compared replacement ratios calculated with all plots of the study area with replacement ratios calculated with the plots of biogeographical regions (Fig. 1; Gonseth et al., 2001) and analysed the overlap of the uncertainty ranges.

(4) The relationship between patch size and number of species can be described with a power function (Arrhenius, 1923; Review in Dengler (2009)). To analyse, if patch size might affect our results, we calculated the linear regression lines between the log-transformed patch size and the log-transformed species number for all vegetation types. We tested, if slope significantly differs from zero.

3. Results

We show the replacement ratios with a replacement fraction of 60% for all vegetation type pairs of the dry grassland data set in Fig. 3. The vegetation types are arranged according to their similarity. Five clusters of vegetation types have been identified based on the cluster analysis (indicated on the bottom of Fig. 3). Cluster C1 consists of high-altitude vegetation types; Clusters C2 and C3 consist of low-altitude vegetation types; Cluster C4 is composed of vegetation types that are typical of the inner-alpine biogeographical region *Valais*; and Cluster C5 consists of two very rare vegetation types. Both, the cluster analysis on high-value species and the cluster analysis on all species derive the same five clusters.

The diagonal cells correspond to in-kind compensation. There are 49 pairs with replacement ratios $\leqslant 5$. The 11 vegetation type pairs with small replacement ratios ($\leqslant 3$) are concentrated in Clusters C1, C2, and C3. These vegetation types represent most of the patches outside of the *Valais* biogeographical region.

The selection of biodiversity surrogates influences the replacement ratios. Fig. 4 shows the replacement ratios calculated with the two different species sets (all 466 plant species and the 358 species of high conservation value). The replacement rations calculated with the 358 high-value species are on average 11% smaller than the replacement ratios calculated with all 466 species. The difference is especially high for the replacement of low-value vegetation types through high-value vegetation types (on average 31%; indicated with triangles in Fig. 4).

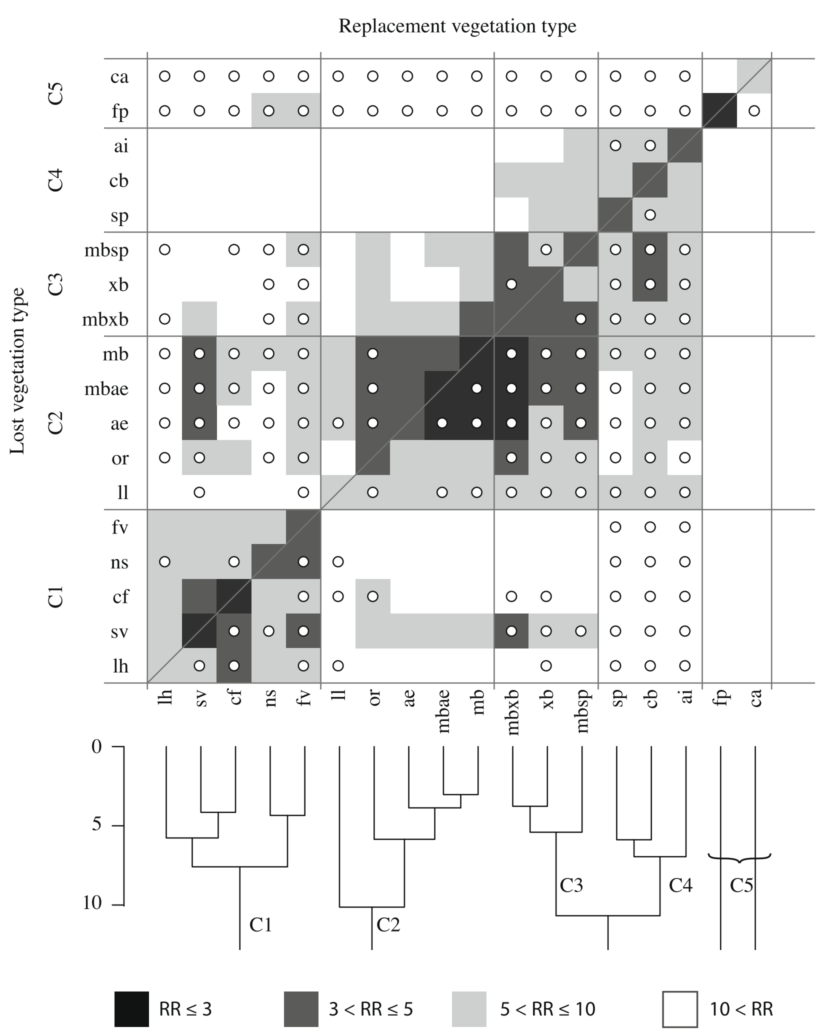

Fig. 3. Replacement ratios (RR) for all pairs formed by the 18 vegetation types. For abbreviations, see Appendix A. Dots mark the lower replacement ratio of the vegetation type pairs built by swapping lost and replacement vegetation type. For example, the replacement of an *mbae*-plot by an *mb*-plot needs a lower replacement ratio then the reverse case. Vegetation types are arranged following the complete linkage dendrogram shown in the lower part of the figure (clipped at the RR-level of about 10). Clusters are marked as C1–C5. Additional information is given in Section 2.2.

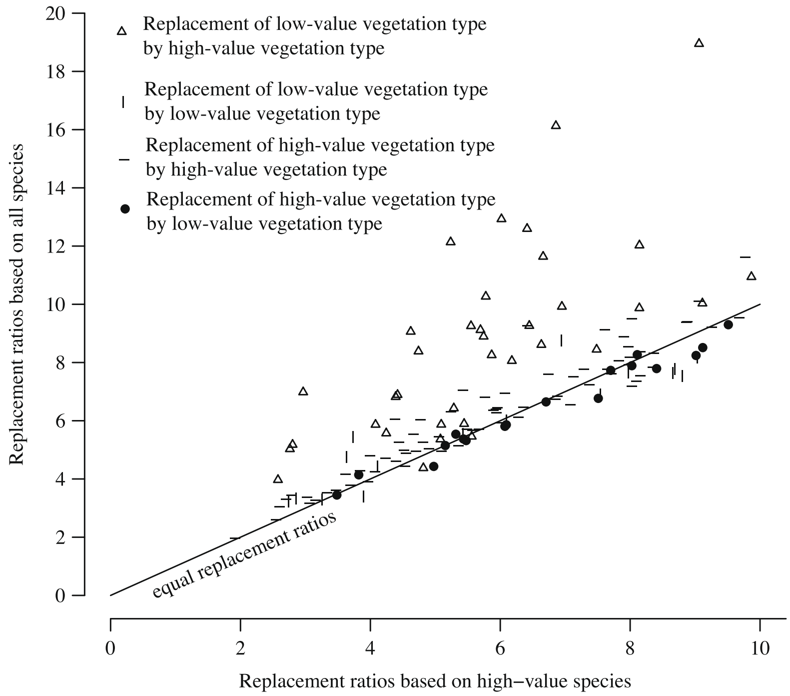

Fig. 4. Replacement ratios calculated on the basis of 386 high-value species were compared with ratios based on all 466 species. Each mark indicates one vegetation type pair. Symbols indicate the classification of the vegetation types pairs according to their conservation value. The figure is clipped to replacement ratios ⩽10 respective to the *x*-axis.

Clearly, replacement ratios are affected by the choice of the replacement fraction. In this paragraph RR_T denotes the replacement ratio calculated with replacement fraction *T*. The slopes of the regression lines (*b* in Table 1, step 4) that were used to compute the replacement ratios varies in the range 0.43 ± 0.07 (mean and standard deviation) for vegetation type pairs with $RR_{60\%} \leqslant 10$. The $RR_{70\%}$-values were 44 ± 9% higher than the $RR_{60\%}$-values, and the $RR_{50\%}$-values were 35 ± 5% smaller than the $RR_{60\%}$-values. For the replacement fraction *T* = 60%, 11 $RR_{60\%}$-values were ⩽3, for *T* = 70% there are no replacement ratios ⩽3, and for *T* = 50%, 42 $RR_{50\%}$-values were ⩽3.

Spatial autocorrelation has a minor effect on replacement ratios. Even for vegetation types with a significant autocorrelation (level 0.05), the decrease of the species replacement fraction is below 0.1 per 100 km (exception: *Arrhenatheretalia*). The 50% uncertainty range of the values for biogeographical regions does not overlap with the 50% uncertainty range of the national data. On the 35%-level, there is an overlap only for three pairs involving *Festucion variae* and *Seslerion albicantis*.

The influence of patch size on replacement ratios is probably negligible compared with the effects of the species set and the effect of the replacement fraction. Linear regression between the patch areas and the number of species in their sample plots – both log-transformed – results in lines with a positive slope for half of the vegetation types (significance level 5%, $R^2 < 0.1$). If patch area is doubled, there are approximately 6% more species. However, a simulation study supports the hypothesis that this correlation does not translate into replacement ratios. (Results not presented here.)

4. Discussion

4.1. Replacement ratios

The estimated replacement ratios for Swiss dry grassland biotopes vary considerably and can attain extremely high values. We focus in this paragraph on a comparison with Righetti's (2002) values because they are currently official guidelines for replacement projects in Switzerland. Righetti based his estimates on expert judgement and allowed for complementarity (Kati et al., 2004) and therefore used a very different method and more lenient assumptions. For very similar vegetation types (e.g. different variants of *Mesobromion*), our replacement ratios based on a replacement fraction of 60% are only slightly higher than Righetti's (2002) values (e.g. Righetti suggested replacement ratios of approximately 1.5 while our estimates are 3 ± 0.5). For a replacement fraction of 50%, we get approximately the same replacement ratios as Righetti (2002). However, for compensation with vegetation types that are less similar (e.g. a *Xerobromion* by a *Mesobromion*, our estimates of 8.2 are much higher than Righetti's suggestions of 1–2.5. For replacing high quality dry grassland with low quality orchard Righetti suggest a replacement ratio of 3.5, where with our assumptions we would get probably a ratio over 200 because no high-value species of dry grassland would be expected in an orchard. It is our feeling, that replacement ratios >5 will generally not be possible in conservation practice in the context of the Swiss dry grassland biotopes. Based on our results the potential for replacement compensation in the Swiss dry grassland biotope network seems therefore rather limited.

The following example shows how the estimated replacement ratios can be used to determine replacement compensation in a fictitious problem also discussed by Righetti (2002). We assume that a small road is built through a dry grassland patch, so that 0.2 ha of *Mesobromion* is destroyed and must be compensated. It is suggested to compensate by enlarging a *Mesobromion*-patch and by generating a fringe association along the edge of a woodlot nearby. According to the calculations, the replacement of *Mesobromion* by *Mesobromion* needs a replacement ratio of 2.7, and the replacement of *Mesobromion* by fringe needs a replacement ratio of 3.8 (Fig. 3). Consequently, the loss would be balanced by newly created 0.27 ha of *Mesobromion* and 0.38 ha of fringe.

Though plot-based replacement ratios can be used in practice for estimating compensation requirements in a dry grassland reserve network, they replace by no means a detailed conservation planning. Indeed, our replacement ratios should be understood as a rough estimate of the compensation needs. For thorough planning, guidelines, compensation goals and an analysis of the local biodiversity potential are indispensable. The pursuit of other targets (e.g. the mitigation of fragmentation effects, the recognition of the needs of Red List species, the consideration of regeneration time and other biodiversity conservation goals for the reserve network) is facilitated by the fact that the replacement area always is considerable larger than the lost area.

The estimated replacement ratios for Swiss dry grassland biotopes have several characteristics which warrant further discussion. Clearly, the more similar two vegetation types are, the smaller is their replacement ratio. For in-kind compensation, replacement ratios are therefore relatively small. Furthermore, small replacement ratios are typical for tightly defined vegetation types (e.g. the *Mesobromion* variants or the high-altitude vegetation types *Seslerion albicantis* (*sf*) and *Caricion ferrugineae* (*cf*), Fig. 3) and for vegetation types represented in our data set by a very small number of sampling plots such as the *Festuca paniculata* swards. Groups of vegetation types with relatively small replacement ratios among each other represent rather well distinct altitudes and bioregions. Indeed, vegetation types of low altitude, of high altitude and of the bioregion *Valais* are in the same cluster, respectively.

An ecotone association such as the dry fringe (*Origanethalia*) shares its species list with several hotspot associations (e.g. *Mesobromion* variants, high-altitude vegetation types, vegetation types of the *Valais*). The replacement of hotspot associations with ecotone association has therefore relatively small replacement ratios. For instance, one needs about four plots of dry fringe to replace one patch of the typical *Mesobromion* with a replacement fraction of 60%. Therefore, reserve network design should favour hotspots (Brooks et al., 2001). This discussion on ecotones and hotspots further illustrates the importance of the biodiversity surrogate choice. If instead of vascular plants insects or spiders would have been used as surrogate, the dry fringe would be a hotspot. Consequently, only few other vegetation types would be able to replace it with low replacement ratios (Masé, 2005).

Replacement ratios are not symmetrically distributed respective to the diagonal in Fig. 3. If most of the species of a lost plot (e.g. the eutrophic *Mesobromion*, C2 group, Fig. 3) are present in a replacement plot (the oligotrophic *Mesobromion*/*Xerobromion*, C3 group, Fig. 3), the replacement ratio is lower than if most of the species of the replacement plot are present in the lost plot.

The results of the cluster analysis for the two species sets show that the relative values of the replacement ratios are not sensitive to the list of species considered for the analysis (and therefore the biodiversity surrogate). However, Fig. 4 shows, that the absolute values of the replacement ratios are sensitive to changes in the species list. As an example, we look into the replacement of a dry fringe by a *Mesobromion*. If the replacement of 60% of all dry grassland species of the fringe is the target, more *Mesobromion*-plots are needed than if 60% of only the high-value species should be replaced. Clearly, in the second case low-value species present in the fringe must not be replaced (for theoretical explanation see also Appendix B). Therefore it is important that biodiversity surrogates used for replacement calculation correspond well to the conservation goals of the reserve network.

4.2. Conclusion for policy makers

The presented method can be a useful tool for the exploration phase of compensation planning. The method is ideally applied in the following context: (1) Habitat loss is unavoidable or negative effects on biodiversity cannot be mitigated (Cuperus et al., 1999). In these cases most impact regulations (Rundcrantz and Skärbäck, 2003) demand a replacement compensation. (2) Data about biodiversity surrogates (e.g. presence and absence of vascular plants) is available for the affected biotope types. This generally is the case for systematically planned reserve networks. (3) It is accepted that compensation should replace a considerable part of the lost features, but other measures (for example the promotion of threatened species elsewhere in the reserve network) should optimise ecosystem functions and biodiversity conservation at a larger scale. Governmental biodiversity conservation policy needs to define how much should be replaced and which other targets have to be reached. (4) The explorative phase for which our method delivers key numbers is followed by a detailed planning phase, in which field data is collected and a comprehensive set of replacement goals is pursued. (5) A monitoring program is installed to test if biodiversity conservation targets are reached, so that corrective measures can be taken if required.

4.3. Outlook

Future research on the potential of replacement ratios for conservation planning should address the following issues: (1) Typically, replacement projects have strong stakeholder participation (Brooks, 2002; Reed, 2008). Therefore, the choice of the biodiversity measure is crucial as it must be simple enough to gain stakeholder acceptance but complex enough to be significant for conservation. Research should address how biodiversity is measured, how participation is organised and how replacement schemes are embedded in the biodiversity conservation strategy (Allen and Feddema, 1996; Cowell, 2003). (2) From applied restoration ecology research we expect insights into techniques on how to restore dry grassland. These techniques should take into account soil nutrition condition, the establishment of dry grasslands species, and the successional dynamics under agricultural management (review in Young et al., 2005; Butaye et al., 2005; BUWAL, 2006; Stampfli and Zeiter, 1999). (3) From quantitative vegetation science, insights about the influence of the species' spatial distribution on replacement ratios and other characteristic values used in the set covering framework would be helpful. Our data set did not allow a satisfying analysis of such effects and could benefit from the analysis of further data (Palmer et al., 2007; Wagner, 2003). (4) Simulation studies should provide insights about the dynamic of reserve selections ("swapping"; Strange et al., 2006).

To conclude, replacement ratios provide a convincing starting point for specifying replacement compensation in reserve networks. Though the approach presented in this paper seems very promising, more research on replacement ratios estimated directly from ecological data would greatly benefit conservation planning.

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Appendix A

Characterisation of the 18 vegetation types.

Abbrev.	Name[a]	Reference	Number of patches[d]	Value[e]	Altitude[f]	Valais[g]	Cluster[h]
lh	Low diversity, high altitude dry grassland	–	245	0.3	*H*		C1
sv	Blue Moorgrass slopes	*Seslerion albicantis*[b]	1242	**0.6**	*H*		C1
cf	Slopes of Rusty Sedge	*Caricion ferrugineae*[b]	474	**0.6**	*H*		C1
ns	Species-rich Mat-grass swards	*Eu-Nardion*[b]	376	**0.6**	*H*		C1
fv	Species-rich Varicoloured Fescue Garland–grassland	*Festucion variae*[b]	999	**0.6**	*H*		C1
ll	Low diversity, low altitude dry grassland	–	270	0.3	*L*		C2
or	Dry Fringe communities	*Geranion sanguinei*[b]	219	0.4			C2
ae	Dry, species-rich, high-nutrient grassland	*Arrhenatheretalia*[b]	1849	0.0	*L*		C2
mbae	High-nutrient semi-dry grassland	*Mesobromion*[b]	3308	0.4	*L*		C2
mb	True semi-dry grassland	*Mesobromion*[b]	2440	**0.6**	*L*		C2
mbxb	Drier semi-dry grassland	*Mesobromion*[b]	599	**0.8**			C3
xb	Subatlantic dry grassland	*Xerobromion*[b]	291	**0.9**			C3
mbsp	Steppe-like, semi-dry grassland	*Mesobromion*[b]	179	**0.8**	*H*	+	C3
sp	Steppe-like dry grassland	*Festucion valesiacae*[b]	590	**0.9**		+	C4
ai	Semi-ruderal dry grassland	*Artemisio absinthii – Elymion hispidi*[b]	379	**0.6**		+	C4
cb	Subcontinental dry grassland	*Cirsio-Brachypodion*[b]	63	**1.0**		+	C4
fp	*Festuca paniculata* slopes	*Festuca paniculata* swards[c]	5	**0.6**	*H*		C5
ca	Southern Alpine blue Moorgrass slopes	Southern rusty sedge grasslands[c]	3	**1.0**	*H*		C5
		Total	13,531				

[a] Eggenberg et al. (2001).
[b] Ellenberg (1996).
[c] Commission of the European Communities (1991).
[d] Datacenter Nature and Landscape (DNL) at WSL.
[e] See Section 2.1 and Eggenberg, unpublished project report, 17.7.2001. Vegetation types with high conservation value are characterised by values >0.5 (bold).
[f] The altitudial median of all patches is 1150 m asl, *H* means: >66.7% of the patches are at altitudes >1150 m asl, *L* means:>66.7% of the patches are at altitudes <1150 m asl.
[g] + Means: >50% of patches are located in the biogeographical region *Valais* (Fig. 1).
[h] Clusters in dendrogram (Fig. 4).

Appendix B. Effect of the species list composition on replacement ratios

In this annex we explain the effect of the species list composition on the replacement of a vegetation type of low value than vegetation type of high value (triangles in Fig. 4). Vegetation types of low value have in general more species of low value then vegetation types of high value. We analyse the replacement of one (lost) plot by several replacement plots, assuming that the vegetation type of high value has no species of low value. The number of species present both in the replacement plots and in the lost plot is denoted by a. The number of species present only in the lost plot is denoted by b. A reduction of the number of replacement plots causes a decrease of a by x and an equal increase of b. The decrease of b caused by concerning only species of high value instead of all species is denoted by y. The species replacement fraction (SRF) is defined as $s = a/(a + b)$ [1]. Now, we reduce the number of replacement plots and restrict the concerned species list on species of high value, whereby the SRF stays constant. For the SRF, we get $s = a/(a + b) = (a - x)/((a - x) + (b + x) - y$ [2]. Solving equation [2] for x by using [1], we get $x = s * y$ [3].

Example: with $s = 0.60$ (e.g. $a = 30$ and $b = 20$) and $y = 10$ we get with [3] $x = 6$. This means: We assume a SRF of 60% and that 10 of the 20 species only present in the lost plot are of low value, then six species less must be present in the replacement plots. This can be done with less replacement plots. Therefore, for the replacement of a low-value vegetation type by a high-value vegetation type we usually get a smaller replacement ratio than if the replacement ratio is calculated with all dry grassland species.

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