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Draft methodology to analyze whether networks represent adequately habitats and species of Community interests

Deliverable 11 of Work package 4

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1 Contents

2 Summary

Conservation area networks are area sets, aiming to protect biodiversity in those areas. The question how to identify areas of conservation interest should be subject to careful conservation planning, to ensure allocating the scarce resources most efficiently. Pressey (1990) presented an intriguing figure, when he studied development of allocated reserve areas and compared this to the total area needed for protection of biodiversity in New South Wales, Australia (Figure 1). It clearly can be seen that the two lines do not approach each other as desired, but merely reflect on opportunistic reserve site allocation, which does not really lead to an implementation of a successful area network. Such data, unfortunately, is not yet analysed for Europe or any of the European nations. However, as the site selection in Europe does not always follow conservation aspects (see D14), it has to be assumed that also in Europe the two lanes will be parallel rather than approaching each other.

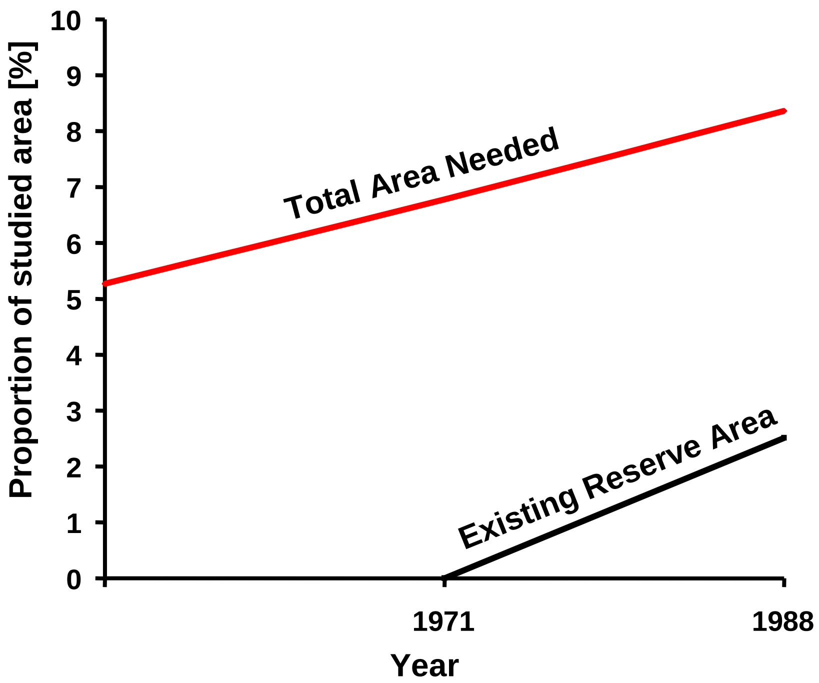

Figure 1 : Accumulation of reserves in north-western New South Wales between 1971 and 1988 (lower line) and the total area needed in 1971 and 1988 to represent all land systems (environmental classes). Re-produced from Pressey (1990).

The NATURA 2000 network aims to fulfil the protection of species and habitats of Community interest on the European scale. To achieve this goal most effectively, it is necessary to follow a scientific sound protocol of area selection, which has been intensively developed in the recent years. This methodological framework for a systematic conservation planning process includes the following steps:

– Delineate manageable, potential priority sites

– Identify the conservation goals of the network
As it is not possible to cover biodiversity in the whole region more specific

goals have to be defined for specific regions (i.e. conservation of rare species, conservation of indicator species, conservation of specific habitat types of processes, assurance of ecosystem services)

– Nominate biodiversity surrogates
  To be able to determine which sites should be protected, knowledge on the state of biodiversity (or the delineated goal of the network) has to be gathered. This cannot be done for complete sets of species/habitat types in all potential areas, therefore surrogates for biodiversity have to be defined (e.g. indicator species/habitats).

– Setting of quantitative representation targets
  To verify the success of the conservation network a clear quantitative representation target has to be defined (i.e. at least three occurrences of each species, 1000 ha of each habitat type)

– Selection of priority sites by a standardize method
  This step includes the determination of principles of site selection. It follows mathematical algorithms that allow for multiple criteria optimization by evaluating a reserve in regard to its role within the context of many other units. Site selection can be based on different features such as complementarity, rarity, adjacency and vulnerability. The algorithms weigh the different feature and aid in selecting areas from potential sites. Important methods are iterative algorithms, simulated annealing and linear programming.

It is important to note that so fare this unified approach has not been undertaken by the members of the European Community. If they agreed on the already elaborated methods of network sites selection, the streamlining of the reporting process for the NATURA 2000 areas would be much facilitate and would further aid to allocate NATURA 2000 sites more efficiently. The NATURA 2000 network still lacks this sound common framework that would allow for an efficient reserve site selection (see e.g. D14). Nevertheless, the framework can be used to identify specific conservation goals and to check the efficiency of the network by testing whether the conservation goals have been reached. For example, the proportion of the amount of habitat needed to the area for the protection of all habitats of Community interest that is needed should decrease with the allocation of more reserve sites.

3 Methodology for analyzing the efficiency of reserve site selection

3.1 General background

The protection of the full spectrum of regional biodiversity is one of the most important challenges for nature conservation in practice (Frankel & Soulé 1981, Vane-Wright 1994, Margules & Pressey 2000, Groves et al. 2002, Specht et al. 2003), and from the perspective of nature conservation alone, one would attempt to have the largest reserve system possible. However in reality, the extent of any reserve system will be limited by socio-political and economic constraints and it won't be feasible to conserve every site that is of some biological interest (Hopkinson et al. 2001, Sarkar & Margules 2002). Consequently, building a reserve network that will conserve biodiversity effectively is not a process of accumulating as much land as possible, but instead biodiversity protection efforts have to be carried out as efficiently as possible. I.e. a set of priority sites has to be identified, which suites best for conserving a wide array of species or other biodiversity features, and similarly minimizing the area required and reducing conflicts with competing land-uses (Pressey et al. 1993, Saetersdal & Birks 1993, Margules & Pressey 2000, Polasky et al. 2000, Faith et al. 2001a, Pressey & Cowling 2001, Sarkar et al. 2002, Anderson et al. 1999, Gaston et al. 2002, Garson et al. 2002, Margules et al. 2002, Williams et al. 2002, Groves 2003). As a consequence of these increasing demands methods for the selection of conservation priority sites have become more sophisticated and systematic within the last two decades. Instead of considering e.g. scenic areas or species hotspots as decisive for conservation decisions, complementarity based approaches are commonly applied to establish reserve networks that assure the representation of a large number of species, habitats, but also of processes that maintain and produce biodiversity in a limited number of sites (Polasky et al. 2000, Cowling & Heijnis 2001, see also D16). By means of such systematic approaches, conservationists also try to escape from solely reacting to a crisis, but instead to act in a timely and proactive manner (Tabarelli & Gascon 2005).

3.2 Necessity for a systematic identification of conservation priority sites

The existing nature reserves were often not selected according to specific biodiversity conservation objectives, but instead they are found in places that are unsuitable for urban or agriculture development, as e.g. in high altitude, steep, or rough areas (Pressey et al. 2000, Rouget et al. 2003b). Besides the mere availability, reserve sites were also chosen for cultural or scenic reasons, as favorite holiday destinations, as game reserves, for water protection, or for the protection of a few charismatic species. Due to this ad hoc-selection (Pressey 1994) most existing reserve systems have an excess of high mountains, rocky infertile soils, salt lakes, or inaccessible swamps; flat, well-drained, and fertile land is rarely conserved (Pressey & Tully 1994, Sarkar et al. 2002). And if conservation goals were explicitly specified, this was mostly done separately for each single site without accounting for the relevance of the respective site for biodiversity conservation in a certain planning region. As a result of such a site selection, regional biodiversity is rather imperfectly represented, because some species, communities, or ecosystems are left without protection, while other features are repeatedly represented (Rebelo & Siegfried 1990, Freitag et al. 1998, Araújo 1999, Cowling 1999, Lombard et al. 1999, Possingham et al. 2000, Sarakinos et al. 2001, Cantú et al. 2004).

Any approaches aiming for an identification of an efficient regional reserve system, based on sites already set aside for conservation require a higher number of sites compared to approaches which were right from the beginning systematically carried out (Pressey & Nicholls 1989a, Lunney et al. 1997). At the same time facing an ever-growing pressure of competing land-uses the availability of nature reserves has become more limited. Therefore, the efficient utilization of conservation resources is more critical than ever before (Pressey et al. 1994, Williams et al. 1996a, Arthur et al. 1997, van Langevelde et al. 2002).

3.3 Definition of priority sites and networks of priority sites

Priority sites for nature conservation can be defined as areas which are at the respective planning level of outstanding importance for the conservation of species or habitat diversity as well as for ecosystem functions (see e.g. Pressey et al. 1993, Bibby 1998). In general priority sites have two functions: They should represent the known biodiversity of the region they are situated in and they should protect biodiversity from processes that threaten its persistence (Margules et al. 2002). As these areas should be scheduled for conservation action first, they are called "priority sites". Since the general intent is to set a basis for biodiversity conservation, according to specified conservation goals (Williams et al. 1996b, Williams 1998), priority sites need not be strictly designated as nature reserves. Instead, appropriate management measures can sometimes be carried out without the need to change the ownership (Margules et al. 2002), provided that nature conservation is given priority over competitive land use forms. In the long run, conservation actions should not be restricted to such priority sites or nature reserves but instead should be broadened by integrating also the surrounding matrix for an achievement of the envisaged conservation goals (Jongman 1995).

In order to overcome the constraints of an ad hoc-site selection, priority sites should no longer be treated as areas being independent from one another, but instead they should be identified from a regional pool of potential priority sites and combined to a reserve network (Williams et al. 1996b). The selection of any planning unit for such a network is carried out by evaluating it with regard to its contribution to regional conservation goals within the context of all other potential priority sites (Bibby 1998, Urban 2000, Sarkar et al. 2002). In most cases various site combinations are suitability for a fulfillment of the conservation goal. Thus, conflicts which may be impossible to solve on individual sites may be resolved on a regional scale. Ideally, such a network will collectively satisfy both, realistic regional conservation targets as well as social and economic criteria (Margules et al. 2002) and thus may represent the ‘backbone’ for nature conservation strategies in a regional context (Williams et al. 1996a, Willis et al. 1996, Margules et al. 2002).

It is generally not possible to search manually for the set of sites that best meets the stated conservation goals. Furthermore, as demands of competing land users increase, the need to choose a network of reserves that will capture the "most" for the least "cost" becomes decisive. Within this framework good guesses may be not good enough, particularly for those user groups whose livelihood depends on these resources. Thus, during the past much effort has been spent on the development of methods for the establishment of conservation networks (e.g. Pressey et al. 1993, Justus & Sarkar (2002). Against this background the following aspects will be dealt with in this deliverable:

- Outline of a systematic biodiversity conservation planning process
- Introduction to standardized site selection methods for reserve networks
- Comparison of selection methods
- Discussion of the applicability of these methods with respect to an implementation in conservation praxis

3.4 Methodological framework for a systematic conservation planning process

The selection of priority sites is essential for a successful implementation of nature conservation measures (Shafer 1999), but it also should be kept in mind that site selection is only one part of a systematic biodiversity conservation planning. In fact the planning process requires various steps in which decisions on the delineation of planning units, the setting of conservation goals and representation targets, the nomination of biodiversity surrogates, the collection of data, etc. have to be made. Moreover, the process is not unidirectional, but there will be many feedbacks and reasons for altering decisions (see e.g. Williams et al. 1996b, Anderson et al. 1999, Pressey 1999b, Shafer 1999, Margules & Pressey 2000, Pressey & Cowling 2001, Gaston et al. 2002, Groves et al. 2002, Noss et al. 2002).

3.4.1 Delineation of the planning region and nomination of planning units

As a first step a planning region has to be identified for which the reserve network should be developed. For this purpose a global (e.g. Mittermeier et al. 1998, Olson & Dinerstein 1998, Rodrigues et al. 2004) or ecoregional approach (Dinerstein et al. 2000, Groves et al. 2000) might be adopted. Likewise, any other spatially defined geopolitical or ecological region might be reasonable.

Within each planning region individual geographically defined units of land or water have to be nominated which will serve as planning units (= potential priority sites). These planning units should represent manageable-sized parts of the landscape and they can differ with respect to size and form. Irregular polygons like habitat fragments, watersheds, or distinct habitats (Ryti & Gilpin 1987, Margules et al. 1988, Pressey & Nicholls 1989a, Bedward et al. 1992, Saetersdal & Birks 1993, Margules et al. 1994, Kershaw et al. 1995, Ranta et al.

1999) may serve as planning units as well as regular grid cells defined e.g. by latitude and longitude (Kirkpatrick 1983, Rebelo & Siegfried 1990, Rebelo & Siegfried 1992, Nicholls & Margules 1993, Freitag et al. 1996, Freitag et al. 1998, Araújo 1999). Planning units may also be delineated based on a combination of the parameters topography, landform, geology, climate, and vegetation (Fairbanks & Benn 2000). The nomination of planning units (grid cells vs. irregular polygons as "real" sites) has to be carried out purposefully. Whereas grids as planning units are often applied on a coarse spatial scale and used for a delineation of search areas ("hotspots") for future conservation actions, real sites predominately play a decisive role on a fine spatial scale and for the identification of sites which are appropriate for the implementation of concrete conservation management. Furthermore grid cells may encompass rather heterogeneous habitat structures and ecotones as they are formally delineated and especially in densely human populated areas a considerable difference between grid dimension and the area available for conservation purpose may emerge (Prendergast et al. 1999).

In any case, the size of the planning unit has to be seen in relation to the reliability of mapping and relative to the scale of the underlying features (Pressey & Logan 1998). If the units are too small, there may be so many that network assembly will be unacceptably slow or the network comprises just small fragments which are not sufficient for area-demanding species. If the units are too large, the analysis will be too coarse and fail to adequately represent reality.

3.4.2 Identification of conservation goals

Conservation goals are not universal, but differ according to the regional characteristics as well as among people and situations. Therefore, conservation goals ought to be made explicit and agreed as broadly as possible at the beginning of any particular site selection process if conflicts are to be minimized (Williams 1998, Noss et al. 2002, Noss 2003).

Usually the representation of the full spectrum of regional biodiversity is considered and agreed as an overall objective which has to be specified for the respective planning region by more detailed conservation goals, as e.g.

- the conservation of rare or endangered species,
- the conservation of indicator species,

- the conservation of characteristic species or characteristic habitat types,
- the conservation of all species or species of selected taxonomic groups
- the assurance of the long-term persistence of populations,
- the assurance of connectivity,
- the conservation of ecological and evolutionary processes,
- the assurance of ecosystem services.

Additionally it may be requested that the selected sites are compatible with requirements resulting from e.g. scientific research, education, passive recreation, and even some forms of resource consumption respectively sustainable use of biodiversity (Frankel & Soulé 1981, Vane-Wright 1994, Faith & Walker 1996c, Shafer 1999, Margules & Pressey 2000, Groves et al. 2002, Margules et al. 2002, Pressey et al. 2003, Specht et al. 2003).

3.4.3 Nomination of biodiversity surrogates

The selection of priority sites is based predominately on a comparison between different sites. Consequently, homogeneous information on biodiversity features for all planning units is essential in either case (Freitag et al. 1996, Freitag et al.1998, Prendergast et al. 1999). However, neither the time nor the resources required to survey all regional biodiversity features will ever be available (Prendergast et al. 1993, Faith & Walker 1996b, Faith & Walker 1996a, Bibby 1998, Freitag et al.1998, Shafer 1999, Polasky et al. 2000, Reyers et al. 2000, Araújo et al. 2001, Gaston et al. 2002, Kintsch & Urban 2002, Williams et al. 2002). Thus, the identification of a representative minimum-set of sites has to rely on biodiversity "surrogates" (Flather et al. 1997, Spector & Forsyth 1998, Fleishman et al. 2000), which can be surveyed in a feasible cost- and time-efficient but accurate manner. Basically these approaches assume that the occurrence patterns of specified surrogates are correlated with the presence or abundance of unsurveyed taxa – even if information on details may be lost, as no single parameter is capable of capturing all biological diversity (Williams & Gaston 1994, Caro & O'Doherty 1999, Margules & Pressey 2000, Reyers et al. 2000, Faith et al. 2001b, Fleishman et al. 2001, Garson et al. 2002, Reyers et al. 2002, Sarkar & Margules 2002, Mac Nally & Fleishman 2004).

The question whether surrogates may be adequate representatives for total biodiversity is up to now not finally answered. And furthermore the decision on which biodiversity features may be used as surrogates is influenced by the availability of a sufficient area-wide data set across the planning region and by the acceptable trade-off between precision and cost of data acquisition. Subsequently a short overview of often applied "surrogates" will be given.

3.4.4 Aggregated attributes

Aggregated attributes comprise e.g. species richness, size of the area, or number of rare or endangered species. As these attributes lack reference to particular species respectively species composition, complementarity can not be taken into consideration when combining single sites to a conservation network. Thus, an application of aggregated attributes may be useful for a preliminary evaluation of the conservation potential on coarser spatial scales, but is probably inadequate as a basis for establishing a reserve network or for decisions on biodiversity conservation on finer spatial scales.

3.4.4.1 Species

For the usage of species groups as biodiversity surrogates, it is imperative to have sufficient taxonomic and biological knowledge and that data on distribution pattern, abundance, or spatial extent can be gathered by field surveys or from collections of field records held in museums or herbariums (Spector & Forsyth 1998, Fleishman et al. 2000, Williams et al. 2002). Basically these approaches assume that areas rich in the specified surrogate species will be indicative of similar trends in unsurveyed taxa (Reyers et al. 2000). Up to now surrogate species pools have often been derived from the application of concepts as e.g. flagship species, umbrella species, endangered species etc. (see e.g. McGeoch 1998, Simberloff 1998, Zacharias & Roff 2001, Roberge & Angelstam 2004).

In this context, it was often demonstrated that a combination of surrogate species nominated according to the principle of e.g. charisma ("flagship" species), endangerment, or rarity result in rather low representation of other species groups (Landres et al. 1988, Andelman & Fagan 2000, Gustafsson 2000). Thus, it may be assumed that many species rich sites do contain just a few or even no rare species (Prendergast et al. 1993). Nevertheless, an integration of flagship species may be decisive for the acceptance of management measures and, thus, for the success of conservation efforts (Williams et al. 2000).

The use of keystone species, which are thought as critical to the ecological function of a community, remains difficult as experiments that lead to their identification are mostly time consuming and expensive to carry out (Simberloff 1998). As an alternative the umbrella species concept was applied for the identification of surrogate species. With the purpose to improve the efficiency of the surrogate approach, i.e. to enhance the portion of species which benefit from a protection of biodiversity surrogates, also spatial and compositional attributes of the landscape, as e.g. connectivity (van Langevelde et al. 2000), were used for the identification of reasonable surrogates. Lambeck (1997), Brooker (2002), and Freudenberger & Brooker (2004) included threatening processes resulting from effects of habitat loss or fragmentation into site selection by identifying threat sensitive 'focal' species. The critical habitat requirements of each focal species are used to define the amount and configuration of habitats that must be present in the planning region. However, the suitability of such surrogate species for an efficient representation of regional biodiversity, as e.g. number of (endangered, threatened) species has yet not exhaustively been tested in different planning regions. Similarly, the target species concept (Walter et al. 1998, Altmoos 1999) is based on strategies which aim at a protection of animal species including their habitats. Thus, a conservation of sufficient large populations of the target species should also guarantee the conservation of species which rely on similar parts of the ecosystem. In order to minimize the danger of misinterpretation, the nomination of a subset of various species covering different spatial scales or life strategies is suggested (Landres et al. 1988, Saetersdal & Birks 1993).

Sometimes also patterns of higher taxon richness were used as surrogates for species richness (Reid 1998, Williams et al. 1997). Williams & Gaston (1994) demonstrated significant positive correlations between the number of families and the number of species. As they used small, well-known groups they recommended large scale assessments before an application of their approach.

Currently, there seems to be an imbalance between the considered "potential value" and well propounded criteria for the identification of surrogate species as indicators for biodiversity and the actual testing of their efficiency as surrogates

(McGeoch 1998), because in many studies the suitability of surrogate species for an efficient representation of target species is assumed but not exhaustively tested (e.g. Lambeck 1997, Spector & Forsyth 1998, Sarakinos et al. 2001, Bani et al. 2002, Coppolillo et al. 2004).

3.4.4.2 Combination of environmental variables

Area-wide, homogeneous information on environmental variables are often easier and cheaper to sample than species data from various taxon groups (Belbin 1993), thus combinations of environmental variables are often used to generate biodiversity surrogates (e.g. Faith & Walker 1996c, Wessels et al. 1999, Margules & Pressey 2000, Nix et al. 2000, Pressey et al. 2000, Faith et al. 2001b, Groves et al. 2002, Sarkar & Margules 2002, Faith et al. 2003, Higgins et al. 2005, Sarkar et al. 2005, Trakhtenbrot & Kadmon 2005). Surrogates are often generated by a classification or intersection of area-wide

- terrain data (surface morphology): information on elevation, slope, relief, or aspect, which can be gathered from topographic maps or Digital Elevation Models
- climate data: information on rainfall, temperature, evaporation etc. which can be spatially interpolated with the help of a Digital Elevation Model
- physical and chemical substrate data: information on geology and soil

Despite their frequent use as biodiversity surrogates, correlations between combined environmental parameters and species occurrence pattern have rarely been tested quantitatively (Sarkar & Margules 2002). Using 50 x 50 km UTM grid cells, Araújo et al. (2001) did find no correlations between environmental variables and the occurrence of amphibians and reptiles. These missing correlations probably are due to the coarse spatial resolution of the grid cells which do not meet the scale of habitat requirements of amphibians and reptiles. In addition, it needs to be considered that present species distribution is also influenced by extinction- or dispersal-processes, by barriers, biotic interactions, or human land-use history (Araújo et al. 2001). In contrast Cowling & Heijnis (2001) and Lombard et al. (2003) found that members of the plant family Proteaceae were rather well represented by grid cells as identified by the intent of a sufficient representation of broad habitat units. Broad habitat units were derived from an intersection of geology, topography and climate. As these broad habitat units are mainly surrogates for plant diversity (Cowling & Heijnis 2001), it appears plausible that vertebrates are represented by this form of biodiversity surrogates only to a minor degree (Lombard et al. 2003). Similarly Trakhtenbrot & Kadmon (2005) found that vascular plants can be better represented by environmental classes than by randomly selected sites. Above all, correlations between environmental surrogates are seemingly scale dependent. Sarkar et al. (2005) demonstrated that at the finest scale (0.01° x 0.01°) environmental surrogates performed no better than randomly selected sites. However, applying coarser grid cells (0.02° x 0.02° - 0.1° x 0.1°) enables a significantly more efficient representation of species.

In contrast to such classification approaches, Trinder-Smith et al. (1996) favored the approach to select reserve sites which reveal a high variety of different environmental gradients, offering simultaneously various different habitats which may support the conservation of regional biodiversity. As vegetation cover can be mapped from satellite imagery with continuously increasing resolution (e.g. Shafer 1999, Jennings 2000) and vegetation types may represent species combinations and their interactions rather well, information on land-cover are applied as biodiversity surrogates to an increasing degree (Sarkar & Margules 2002). Despite the loss of detailed information Mac Nally et al. (2002) found that vegetation units account reasonably well for birds, mammals and trees, but not for reptiles and invertebrates. Additionally, large patches of remaining natural vegetation with large undisturbed core areas often typically harbor more viable populations, offer greater habitat diversity, and support more intact ecological processes (Poiani et al. 2001). Hence, a delineation of land-classes derived from an intersection of both environmental variables and information on land-cover should actually provide valuable surrogates for biodiversity (Anderson et al. 1999, Pressey et al. 2000, Faith et al. 2001b, Noss et al. 2002).

The use of such surrogates as spatial predictors without detailed knowledge of (regional) species occurrences have to be carried out cautiously (Scott & Sullivan 2000). Significant percentages of species may be missed altogether in reserve networks based on habitat classes and those most likely to be missed tend to be rare species or species with restricted ranges Brooks et al. (2004). Thus, in areas where, besides reasonable environmental surrogates, sufficient species data are available it might be recommended to target land classes and available species data simultaneously (Poiani et al. 2001, Kintsch & Urban 2002, Reyers et al.

2002, Lombard et al. 2003). Whereas, in areas where primarily data on only some, e.g. endangered or endemic, species are available, it might be an option, to represent each environmental class in a complementary network, overlay available distribution maps of selected species and add the missing environments to the network (Faith et al. 2001c, Faith et al. 2001b, Noss et al. 2002). Similarly, the location of the selected species could be used as seed points and a complementary network could be built around it (though it has to be taken into account that species-point data may be spatially biased). The integration of different surrogate types supports the achievement of targets for coarse-filter features such as land classes and environmental gradients, and also the identification of areas for protecting viable populations of large herbivores or top carnivores (Cowling et al. 2003). Important sites for the conservation of rare species could be identified and added to the network which otherwise were likely to be overlooked.

3.4.4.3 Collection of information on biodiversity features

Generally, data assembly may comprise the review of already existing data sets or the collection of new (field-) data. Though, in either case the efficiency and performance of reserve systems for biodiversity conservation depends critically on the quantity and quality of the available data basis (Andelman & Willig 2002). Ideally, the type and depth of the data collected should be identical for all surrogates and planning units (Margules & Pressey 2000). Thus, a consistent sampling effort across all planning units has to be assured in order to guarantee that differences in species or habitat composition are due to differences between the individual sites and not caused by differences in sampling intensity (e.g. Freitag et al. 1996, Williams et al. 1996b, Freitag et al.1998, Prendergast et al. 1999, Margules et al. 2002, Williams et al. 2002). However, the establishment of a homogenous data base including reliable absence data is expensive and time consuming, especially for animal species. By making use of area-wide available information on environmental variables, aspects of habitat suitability were given more importance for site selection in order to account for incomplete information on species distribution (e.g. Polasky et al. 2000, Loiselle et al. 2003, Root et al. 2003). Statistical habitat models describe the functional correlation between indicator species and their habitat and support the formalization of correlations between environmental conditions and habitat requirements of the indicator species (Scott et al. 2002, Guisan & Zimmermann 2000). Habitat models are mostly build by the application of Logistic Regression, Generalized Linear Models (Nicholls 1989, Nicholls 1991, Margules & Austin 1994), or Generalized Additive Models (Austin & Meyers 1996). Additionally some "filter cascades" will be applied for model building:

- Correlation of environmental characteristics with the requirements of the indicator species.
- Analysis to what extent landscape configuration conincidences with the movement and distribution pattern of the indicator species.
- Analysis whether biotic interactions possibly restrict the persistence of a population.

Up to now habitat models are predominately applied as surrogates for the respective species itself. But it should also be taken into consideration to broaden this approach und to nominate habitats directly as surrogate for site selection.

3.4.4.4 Setting of quantitative representation targets

Quantitative targets are interpretations of broad conservation goals in spatially explicit, quantitative terms (Cabeza & Moilanen 2001, Margules et al. 2002, Noss 2003). Though the setting of such representation targets bears always the risk of implying a false precision, it helps to clarify conservation goals, arguments can be more readily presented to the general public (Williams 1998), and a control of success is facilitated (Margules et al. 2002). Setting quantitative targets requires an answering of two questions:

How much of a feature is "enough"? This question might be specified by

(i) The setting of targets for e.g. species or vegetation types as the minimum frequency how often single attributes should be represented in the network (e.g. at least three occurrences of each species, 1,500 ha of each vegetation type). Since the application of uniform targets for all attributes may fail to acknowledge that some features need more protection than others, different representation targets can be specified for the single species.

(ii) The setting of targets for the spatial configuration of the reserve network, as e.g. minimum size of sites, connectivity, or distance to roads.

(iii) The identification of constraints, e.g. priority sites should reveal minimal previous disturbance from grazing or logging.

How much does it matter if the envisaged representation target is not reached? The answer requires the setting of "penalties" which result from not achieving the representation target for any attribute (Shoutis 2003).

As knowledge is usually not yet available on the adequate degree of replication needed to attain high persistence without unnecessarily compromising the efficiency of the network (Gaston et al. 2002, Groves et al. 2002), representation targets have to be decided on a case-by-case basis, according to the available information and the specific goals (Margules et al. 2002). Within this context representation targets tend to be rules of thumb and account for normative conditions. Thus, whereas general conservation objectives and goals may have long-term relevance for regional planning, targets often have shorter life spans (Pressey et al. 2003).

3.5 Selection of priority sites with standardized methods

The selection of any planning unit for a reserve network requires an evaluation with regard to its role within the context of many other units. A planning unit with several valuable features on its own may or may not be the best choice overall, depending on the distribution and replication of those features in the study area but also on the already selected sites.

Over the last decade, an explicit framework for systematic conservation planning has emerged (e.g. Margules & Pressey 2000) and conservation planners have developed computer-based approaches to make the site selection process more systematic and more explicit. These approaches respond to the perceived need for reserve site selection to be as efficient or cost-effective as possible, given the competing social and economic demands for land and water. They also address the concern that reserve system design should be repeatable, so that the reserve systems can be readily re-evaluated and adjusted over time as conditions change and new information is acquired. Within this context two slightly different aspects of the same problem can be distinguished (Pressey & Cowling 2001):

(i) Review of existing conservation areas and evaluation of the extent to which quantitative targets for representation and design have already been achieved (Gap analysis, e.g. Flather et al. 1997, Araújo 1999, Shafer

1999, Jennings 2000, Rodrigues et al. 2004). This is basically done by comparing distribution of species, vegetation types etc. of interest with the distribution of conservation areas. Within this context established conservation areas can be seen as constraints or focal points for the design of an expanded system.

(ii) Establishment of a "new" network of priority sites where already existing sites do have only minor influence on the selection process.

3.6 Principles of a systematic priority site selection

The application of explicit criteria for reserve site selection instead of intuitive judgments was already demanded by Ratcliffe (1971). And it was also Ratcliffe (1977) who established a list with ten selection criteria: size, diversity, naturalness, rarity, fragility, typicalness, recorded history (scientific records), position in an ecological / geographical unit, potential value, and intrinsic value (see also Justus & Sarkar 2002). A few years later Margules & Usher (1981) distinguished between criteria which can be assessed by single site visits (diversity, size), which require extensive surveys (rarity, naturalness, representativeness), and which require case histories (recorded history, potential value, and ecological fragility). Besides the development of advanced evaluation criteria, Plachter (1991) recommended a combined application of criteria which refer to population and community aspects, to habit structure, abiotic parameter, but also to aspects of landscape structure. And already Kaule (1991) pointed out that both the detection and the analyses of threats is decisive for an evaluation of conservation priority sites. Nowadays, systematic site selection for nature conservation is basically characterized by principles of representativeness, complementarity and efficiency, flexibility and irreplaceability, vulnerability, persistence and viability.

3.6.1 Representativeness

The principle of representativeness aims at the identification of a set of sites which ideally represents the full range of biodiversity (Noss 1990). This includes a consideration of composition (habitat-, species-, and genetic diversity), structure (physical – abiotic – environment), and function (ecological and evolutionary processes, e.g. reproduction, recruitment, migration, or interaction between species).

3.6.2 Complementarity and efficiency

Taking into account that nature conservation is an integral part of overall land use planning and related to the need to obtain maximum biodiversity for any given level as efficient or cost-effective as possible, representativeness was soon accompanied by the principle of complementarity (Bedward et al. 1992, Pressey et al. 1993, Vane-Wright 1994). Basically, complementarity can be described as an approach of adding sites to a network in such a manner that the number of new features is maximized and unnecessary duplication minimized. As the selection of each new site depends not only on its attributes, but also on the combination of features in the previously selected sites (Vane-Wright et al. 1991), the site contributing most will not be necessarily the species richest one (Pressey et al. 1993, Margules et al. 1994, Faith & Walker 1996b, Bibby 1998). Consequently, the conservation value of a site is dynamic and will change as the reserve system is established (Margules & Pressey 2000). Especially in fragmented landscapes, it has to be taken into consideration that also smaller fragments may play a decisive role in improving the efficiency of complementary sets (Tscharntke et al. 2002, Gaston & Rodrigues 2003).In general, however, such fragments are not just subsamples from larger areas, but capture a much greater habitat- and habitat-environment heterogeneity if combined to a series of small- or medium sized fragments than just one large fragment. As a consequence several small fragments might support more species, respectively more endangered species, than the same area, composed of only one or two larger fragments (Schwartz & van Mantgem 1997, Tscharntke et al. 2002, see also Deliverable 16).

Complementarity was explicitly introduced by Vane-Wright et al. (1991), referring to Kirkpatrick (1983) and Margules et al. (1988), who applied complementarity based iterative algorithms with the intent to represent the full range of species and specified habitat types within a planning region. Margules et al. (1988) already distinguished between a selection process, which emphasizes rarity ("select all wetlands with any species which occur only once") respective richness ("select the wetland from each habitat type which has the greatest number of plant species"). This algorithm was improved by Pressey & Nicholls (1989a) by including an additional step in order to select the smallest site from a range of equivalent alternatives. Later on Nicholls & Margules (1993) integrated the aspect of adjacency, in order to choose the nearest site to the already selected ones, provided that sites are otherwise equal with respect to the principle of complementarity.

3.6.3 Flexibility and irreplaceability

Parallel to the principle of complementarity and efficiency aspects of flexibility and irreplaceability were incorporated into a systematic selection process. Flexibility arises from the non-unique occurrences of many biodiversity features and refers to different but equally suitable site combinations, which guarantee the achievement of the respective conservation goal (Pressey et al. 1993, Pressey et al. 1994, Pressey 1999a, Ferrier et al. 2000). Consequently, flexibility will decrease when sites with rare features are lost. Similarly irreplaceable sites can be identified as those sites which are needed in any case to realize the specified conservation goal. The irreplaceability of a site can be defined either as the potential contribution of a site to a conservation goal, or as the extent to which the options for conservation are lost, if the site is lost (Pressey et al. 1993, Pressey et al. 1994). Areas with total or high irreplaceability may become the nodes of an expanding reserve system (Ferrier et al. 2000).

3.6.4 Vulnerability and threats

Vulnerability was defined by Faith & Walker (1996c) as the degree to which a site, in the absence of action, is at risk of loosing its biodiversity. In this context vulnerability takes into account that e.g. some substrates are more in danger to be changed to agricultural land, or some species are more sensitive to fragmentation (Rouget et al. 2003c, Henle et al. 2004a). Hence, it is a measure of threat resulting from current and likely future disturbances, such as e.g. ongoing habitat loss but also invasion of alien species (Lunney et al. 1997, Cowling 1999, Cowling et al. 1999).

Though habitat loss due to land use practices is probably one of the threatening processes simplest to monitor and to predict (Gaston et al. 2002), it has hardly been incorporated in conservation decisions (Rouget et al. 2003c). This deficit is probably due to the fact that spatial dimensions of future land transformations have to be predicted as probabilities (Sarkar et al. 2004), which requires not only a consideration of current but also an analysis of former land-use pattern. As a straight forward approach it could be considered to carry out a "pre-selection" and exclude those sites from the selection process that are susceptible to foreseeable threats or where the implementation of restoration measures is unlikely (Bedward et al. 1992, Williams 1998).

3.6.5 Persistence and viability

Several empirical studies revealed that reserve networks based only on presence / absence data and a search for the minimum area such that all features are represented at least once do not guarantee the long-term viability of populations or maintenance of biodiversity features (see e.g. Rodrigues et al. 2000c, Rodrigues et al. 2000b, Gaston et al. 2002).

As a first approach to enhance the probability of persistence and to represent intraspecific variation either the application of a multiple representation target (Williams 1998, Rodrigues et al. 2000c, Gaston et al. 2002) or an increase of the threshold of the minimum area of sites taken into consideration for a reserve network was suggested (Rodrigues et al. 2000b, Cabeza & Moilanen 2003). The adequate target for each species has to be decided on a case-by-case basis according to the information available and the specific goals established for the network (Rodrigues et al. 2000b). Further, an increase of the representation target will result in a decrease of efficiency and flexibility (Altmoos 1999, Reyers et al. 2002). Considering this background, it was also suggested to support species protection at those sites at which they were recorded with higher abundances (Pressey et al. 1996, Williams 1998, Cowling 1999, Araújo & Williams 2000) or to exclude sites from the selection process, where species viability seems likely to be poor (Gaston & Rodrigues 2003).

Including information on permanence rates (Rodrigues et al. 2000b) or estimations of probabilities of persistence (Williams & Araújo 2000, Araújo et al. 2002a) in the selection process would be an even better strategy. However, knowledge on population viability or spatially explicit metapopulation models (Akçakaya 2000) are currently available only for some single, well-studied species (Lindenmayer & Possingham 1996, Henle et al. 2004b, Gruber 2005).

Based on the results of numerous studies carried out within the last decade it became obvious that an integration of the principle of persistence into site selection process does not only require a consideration of biodiversity patterns, but also an understanding of ecological and evolutionary processes that maintain and generate biodiversity. However, this would lead directly to the challenge of (i) how relevant ecological processes could be identified and (ii) how they could be included in a selection process (Williams 1998, Cowling 1999, Nicholls 1999, Virolainen et al. 1999, Araújo & Williams 2000, Margules & Pressey 2000, Possingham et al. 2000, Cabeza & Moilanen 2001, Gaston et al. 2002, Groves et al. 2002, Margules et al. 2002, Cabeza & Moilanen 2003, Cowling et al. 2003, Rouget et al. 2003a, Rouget et al. 2003b). Up to now ecological processes that are assumed to maintain or generate biodiversity have been identified and delineated primarily as spatial components on rather broad spatial and temporal scales (see e.g. Cowling et al. 2003, Rouget et al. 2003a). Six spatial components have been identified: edaphic interfaces, upland-lowland interfaces, sand movement corridors, riverine corridors, upland-lowland gradients, and macroclimatic gradients. Hence, without a more detailed and generalized evaluation and testing of the underlying assumptions it is fairly subjective to set targets for size, shape, configuration, etc. for such spatial components without knowing the relations to occurrence or abundance of specified species.

In summary, the trade-off between efficiency and persistence will be difficult to minimize, because an efficient site selection may result in reserve networks which are not robust over time while similarly focusing on persistence may result in a biased, inefficient distribution of conservation resources among biodiversity features, some being highly protected while others are quite underrepresented (Rodrigues et al. 2000c, Rodrigues et al. 2000b). Thus, it might also be taken into consideration to establish a reserve network primarily on the principle of complementarity and adding in a second step at those sites which were identified as relevant for metapopulation dynamics of single specified species.

3.7 Systematic site selection methods

In order to assure representativeness, efficiency, flexibility, and persistence various methods for a systematic selection of priority sites are available (see e.g. Scott & Sullivan 2000, Cabeza & Moilanen 2001).

Complementarity based methods

- Iterative algorithms
- Simulated annealing: iterative stochastic global search method which seeks for solutions by an application of intelligent randomization
- Optimal solutions or linear programming (branch-and-bound methods)

Scoring methods

– Simple ranking based on a single criterion

– Multicriteria optimization

Whereas iterative and optimal methods evaluate the attributes of an individual site in the context of all available planning units, simple ranking methods compare the value of a single planning unit to relative, absolute or threshold values. Ranking and optimal methods simultaneously consider all potential sites, whereas iterative methods consider individual sites sequentially.

3.7.1 Iterative algorithms

Basically, iterative algorithms have been applied to the site selection problem to avoid choosing large numbers of areas that merely duplicate the representation of widespread species. They originate from an attempt to automate the process of reserve selection by copying the way in which a person might choose reserves ‘by hand’. Iterative algorithms start with an empty reserve system and choose sequentially the site which adds the most underrepresented attributes (e.g. species) to the set that has already been selected until specified representation targets for each biodiversity feature have been satisfied to the extent possible. As the selection of each new site depends not only on its attributes but also on the combination of features in the previously selected sites, the conservation value of a site is dynamic and will change as the reserve system is established (see e.g. Pressey et al. 1993, Margules et al. 1994, Willis et al. 1996, Csuti et al. 1997, Pressey et al. 1997). The iterative algorithms can be followed by a backward check in order to make sure that none of the planning units selected in an earlier stage have been rendered superfluous by later additions (Bibby 1998, Aggarwal et al. 2000). By including an adjacency constraint a spatial clustering of reserve sites can be achieved (Nicholls & Margules 1993).

There are a few main types of iterative algorithms (richness, rarity, irreplaceability) that are based on a sites by species (biodiversity features) matrix (Freitag et al. 1997, Pressey et al. 1997, Pressey et al. 1999). Examples for the application of iterative algorithms for establishing a conservation network can be found (e.g.) in: Kirkpatrick (1983), Margules et al. (1988), Pressey & Nicholls (1989a), Rebelo & Siegfried (1992), Nicholls & Margules (1993), Kershaw et al. (1995), Freitag et al. (1996), Trinder-Smith et al. (1996), Williams et al. (1996a), Williams et al. (1996b), Willis et al. (1996), Csuti et al. (1997), Freitag et al. (1998), Pressey & Logan (1998), Cowling (1999), Lombard et al. (1999), Pressey (1999a), Richardson & Funk (1999), Clark & Slusher (2000), Williams et al. (2000a), Sarakinos et al. (2001), Sarkar et al. (2002), Siitonen et al. (2002). A comparison of the efficiency of a variety of iterative selection algorithms which relied either on presence-absence representation targets or which intended to represent a minimum percentage of the total area of each land type was carried out by Pressey et al. (1997).

3.7.1.1 Richness-based iterative algorithms

Richness-based algorithms start with the site containing the most species (attributes) and sequentially add the site with the most unrepresented features. When two or more sites add the same number of additional species, several procedures can be used to break ties (Csuti et al. 1997). This can be done with a random choice, taking the first site in the list (Pressey et al. 1997), looking for the site nearest to another already selected (Nicholls & Margules 1993), or applying rarity criteria. Richness based algorithms have the advantage of making immediate contributions to the successful representation of all conservation features and they work reasonably well (e.g. Margules et al. 1994, Pressey et al. 1996, Pressey et al. 1997, Wessels et al. 1999). Richness algorithms are also often called "greedy" algorithms because they greedily attempt to maximize the rate of progress towards the representation target at each step.

Example of a richness algorithm (Margules et al. 1988):

(i) Select all sites which contain species which have a frequency in the data matrix less than, or equal to, the required level of representation.

(ii) Select from the currently unselected sites in which the next rarest under-represented species was recorded that site which will add the most additional species or add the most to the representation of those currently under-represented.

(iii) Where two or more sites contribute an equal number of additional species, select the site with the least frequent group of species. The least frequent group of species is defined as that group having the smallest sum of frequencies of occurrence in the remaining unselected sites.

(iv) Where two or more sites contribute an equal number of infrequent species, select the first site encountered.

3.7.1.2 Rarity-based iterative algorithms

Rarity algorithms are based on the concept that a reserve system should be designed which ensures that the relatively rare conservation features are reserved first. From there a complementary set focusing on the remaining, more common conservation features is build (Margules et al. 1988, Nicholls & Margules 1993, Csuti et al. 1997, Pressey et al. 1997). The rarity of a conservation feature is calculated as the total amount of it across all planning units. As the rarities for various conservation features could be of different orders of magnitude most of the algorithms use the ratio of the number of all available planning units divided by the frequency of a conservation feature (Rarity score = number of sites / frequency of a single conservation feature). Despite some slight differences most rarity algorithms are composed of similar rules:

- **Maximum rarity**: Select the site with the rarest species (= species with the highest rarity score)
- **Summed rarity**: Select the site with highest summed rarity (= sum of rarity scores of each site)
- **Average rarity**: Select the site with the highest average rarity (= sum of rarity scores of each site / number of species of a single site).

Example for a rarity based iterative algorithm:

(i) Select all sites with any species which occur only once

(ii) Starting with the rarest unrepresented species, select from all sites on which it occurs, the site contributing the maximum number of unrepresented species

(iii) Where two or more sites contribute an equal number of additional species, select the site with the least frequent group of species. The least frequent group of species is defined as that group having the smallest sum of frequencies of occurrence in the remaining unselected sites

(iv) Where two or more sites contribute an equal number of infrequent species, select the first site encountered.

3.7.1.3 Irreplaceability-based iterative algorithms

The idea of irreplaceability captures some of the ideas of rarity and richness in a new way (Pressey et al. 1994). Based on the algorithm of Margules et al. (1988) irreplaceability works by evaluating to which extent each planning unit is required to achieve the representation target for a given conservation feature. If the representation target is as large as the total occurrence of the biodiversity surrogate then every planning unit which holds this attribute is necessary for the reserve system. Furthermore, irreplaceability can be defined as the extent to which the options for achieving the targets are reduced if a site is unavailable for conservation (Ferrier et al. 2000). Since the irreplaceability of an area must refer to specified representation targets and the extent to which other sites contribute to satisfying them, complementarity is a key component of the concept of irreplaceability (Pressey 1999a, Ferrier et al. 2000). As irreplaceability is not binary, but ranges from 0 – 1, the results provide a meaningful basis for negotiations between different land users (Pressey 1998).

3.7.2 Simulated annealing

An overall goal of efficient priority site selection is to identify a set of sites that achieves a given level of biodiversity representation with minimum economic cost. The cost may correspond to the suitability of areas for other land uses e.g. forestry or agriculture. However, the "cost" of a reserve network depends on a lot of different parameters and is often not directly available. Thus, it is necessary to apply surrogates for the monetary cost in order to be able to compare numerous portfolios (= subset of selected planning units presently included in the reserve network) for identifying the most efficient solution (McDonnell et al. 2002). When costs are taken into account an area is justified for the priority network only if its complementarity value exceeds its weighted costs (Faith & Walker 2002). Thus, it might turn out that the benefit for getting a conservation feature to its target is outweighed by the cost of the planning unit which it would have to add.

Simulated annealing is a method which offers the possibility to integrate economic cost into the site selection process. For this purpose the following monetary surrogates can be combined to a "cost-function" (Ball & Possingham 2000, McDonnell et al. 2002, Shoutis 2003):

- **Cost of each planning unit**: the most straight-forward approach to set a cost for each site would be to use their respective area as surrogate for monetary costs, or the number of sites, if all sites have equal areas, e.g. grid cells. Though the cost of each of the sites within the reserve system may also represent a relative value, taking into account that some sites may be more difficult or 'expensive' to set aside than others.
- **Penalty cost**: A penalty might be given for not adequately representing a conservation feature. E.g. if the requirement was to represent each conservation feature at least once, the penalty for not meeting the representation target of a given conservation feature would be the cost of the least expensive planning unit which holds an instance of that conservation feature. Initially when no sites are reserved, the penalty for each conservation value is at its maximum value. As sites are reserved, the number of adequately represented biodiversity features increases and the penalty for that conservation value decreases until the target has been reached and the penalty becomes zero. By applying penalties trade-offs between the protection of biodiversity and its equivalent costs can be figured out.
- **Cost of planning units shared boarders**: "Boundary" represents the length (or possibly cost) of the boundary surrounding a reserve system. Thus, given a reserve system of any fixed total size, the lower the boundary length, the more compact and less fragmented the reserve system.

The objective of the site selection procedure would be to minimize the total cost of the reserve network by minimizing the sum area, the boundary length, and the penalty costs. The cost can be measured in any unit, missing conservation feature penalties will be in the same units. Furthermore the components of the cost-function can be modified:

- **Boundary length modifier (BLM)**: The two objectives – minimizing boundary length and minimizing the total area of the reserved sites – must be traded off against each other. Thus, the "cost" of adding a site to the reserve system depends on the spatial relationship between planning units and those already reserved. In order to minimize boundary length and area the BLM is introduced. The constant BLM will be used to weight the cost of the boundary length compared to the cost of area alone (= relative cost of a reserve's perimeter). By varying BLM, the relative importance of compactness and size can be changed. If BLM is set to zero, then boundary length is ignored, increasing the modifier value gives relatively greater importance to boundary costs, multiplying the boundary length with the BLM. A simple measure of the degree of clustering among sites in a reserve system is the total boundary length of the reserve system divided by the area. The BLM is an arbitrary parameter that must be arrived at thorough experimentation. Decisions on the weight which could be given to boundary length or adjacency can be e.g. influenced by the aspect to reduce edge effect or that it is easier to manage a few large areas instead of dispersed fragments (Possingham et al. 2000).

– **Conservation feature penalty factor (CFPF)**: This weighting factor determines the relative importance for adequately reserving a particular conservation feature. The site selection procedure will try harder to meet representation targets for features with higher penalty factors than for those with lower factors. The factor should be the same for most features but lower for those which are of minor importance. Setting it higher than 1 will favor solutions that perfectly represent that feature.

The cost-function is designed so that the lower the value of the cost function the better is the reserve. In its simplest form it is a combination of the total costs of the sites and a penalty for not meeting the representation targets.

$$\boxed{Total\ Cost = \sum_{Sites} Cost \ + BLM \sum_{Sites} Boundary \ + \sum_{CostValue} CFPF\ x\ Penalty + Cost\ Threshold\ \ Penalty\ (t)}$$

Simulated annealing is an optimization method. According to the analogous physical process of heating and slowly cooling a substance (e.g. glass or steel) to obtain a crystalline structure the parameters "temperature" and "time" are used to specify whether a certain neighboring solution to a current solution should be accepted or not (= acceptance function). The simulated annealing algorithm performs many iterations, starting with a randomly generated portfolio, with each planning unit having an equal probability of being selected, and evaluated in how far it meets the required representation target and which costs it produces. On that basis, another portfolio can be generated by making small random changes and accept them only if the changes made an improvement to the costs. This improved set is carried forward to the next iteration (Andelman & Willig 2002). However this algorithm will get stuck when it reaches a point where all small changes will result in higher costs, but a better solution exists elsewhere (= local optimum, Shoutis 2003). In order to avoid being stuck at an local optimum, even changes that increase the cost of the set of the sites may be carried forward at the start of the procedure, to provide the possibility to examine a greater number of different site combinations. During the ongoing iteration process the chance of accepting a bad change are made progressively smaller and at the end only changes that improve the solution are accepted.

In already existing selection programs (e.g. MARXAN, Ball & Possingham 2000), various options are implemented to explore the advantages and disadvantages of numerous network configurations:

- The network can be restricted to a specified cost and modified boundary length. A penalty will be given if the threshold value is exceeded. A severe penalty will result in a preference of accepting shortfalls in meeting representation targets instead of exceeding the threshold.
- Spatial aspects are taken into account by specifying a minimum size before a site will be considered as priority site. It can be set to zero if viable populations of the (surrogate) species can be maintained in small patches. If the threshold is set greater than zero, then only clumps of planning units greater than the minimum size will count towards a specified representation target.
- Spatial aspects can also be taken into consideration by specifying a minimum separation distance: Though generally clustering of sites may enhance the long-term persistence of species by allowing dispersal and colonization of adjacent sites (Önal & Briers 2002), it is also possible to envisage situations where the close proximity of sites would be undesirable, such as spatially correlated environmental fluctuations or the spread of disease between sites (risk of "putting all eggs into one basket"). The aim of setting a minimum separation distance is to spread the occurrences of the biodiversity feature geographically, not to encourage fragmentation.

It is also possible to set the conservation status of each planning unit, which means e.g. that those particular planning units are fixed in the initial portfolio and can not be removed.

Though simulated annealing does not guarantee to find the "best" solution (Andelman et al. 1999, Possingham et al. 2000), but a range of reasonably good solutions to the same problem can be identified by carrying out numerous repetitions of the selection process.

3.7.3 Linear programming

From a mathematical viewpoint, optimal site selection algorithms are those that identify the smallest set of sites, in terms of number of sites or total area, needed to meet a representation target of biodiversity features in a planning region (Pressey et al. 1996).

The problem of obtaining the global optimum is formally identical to the Maximum Covering Problem. Standard integer programming algorithms such as branch-and-bound schemes are often used to find an optimal solution (see e.g. Underhill 1994, Church et al. 1996, Arthur et al. 1997, Pressey et al. 1997, Önal 2003, Önal & Briers 2002). For solving this problem it can be formulated within a decision-theory:

Let the total number of sites be *m* and the number of different species (or other attributes such as vegetation types) to be represented be *n*. The information about whether or not a species is found in a site is contained in a site-by-species (*m* x *n)* matrix **A** whose elements $a_{ij,}$ are a measure of the representation of feature j in each site i:

a_{ij} = 1 if species j occurs in site i – otherwise 0

For each feature j the user sets a minimum level of representation (r_j), that determines whether or not a site is included in the reserve, as the vector **X** with dimension *m* and elements x_i given by

x_i = 1 if the site is included in the reserve site system; 0 otherwise; for j = 1, ..., m.

With these definitions, the minimum representation problem is:

minimize $\sum_{i=1}^{m} x_i$ c

subject to $\sum_{i=1}^{m} a_{ij}\ x_i \geq r_j$, for j = 1, ..., n (subject to each species being represented at least once).

x_i = 0 or 1 for i = 1...m where x_i are the control variables such that x_i = 1 if site i is in the reserve system; x_i = 0 if site i is not in the reserve system for i = 1...m.

The above model maximizes the number of species represented by the selection of a fixed number of areas for a reserve system.

3.7.4 Multicriteria optimization

A "typical" site selection situation is mostly characterized by conflicting goals (several ecological goals as well as socio-economic goals). Thus, an action that performs well in one goal may be poor in another goal, and another action may have the opposite effect. It is obvious that the preferable solution depends first of all on the stake-holder (Drechsler 2004). In this context, a multi-criteria analysis, e.g. the outranking method (Bana e Costa 1990, Drechsler 2004), can be applied for the purpose of priority site selection. The outranking method requires a pairwise comparison of all actions which are taken into consideration for an achievement of the overall goal. The action that leads to the better result is assigned a certain number of 'points'. For each criterion a so-called preference matrix is established that tells for each pair of actions which one is preferred and how strongly it is preferred. These matrices are aggregated and evaluated to rank the actions (Drechsler 2004).

Besides goal conflicts conservation decisions are also constrained by uncertainty. Uncertainty in the decision problem is often caused by scarcity of information needed to predict the consequences of management actions and to mediate the conflicts between conservation and other ecological or socio-economic goals. The main problem of uncertainty for a decision maker is that it reduces his ability to compare two actions and decide which one is better. Thus, besides the consideration of different preferences and a support of the search for compromises, a decision analysis should also be able to consider uncertainties and be compatible with results provided by conservation biological models (Drechsler 2004).

3.7.5 Ranking methods

When applying simple ranking methods the planning units are given scores with respect to various criteria (e.g. diversity, size, summed rarity, cost, etc.). The planning units are ranked only according to one criterion. Highest priority is given to the planning units with the highest score. The values of criteria of a single planning unit can also be compared to relative, absolute or threshold values

4 Comparison of selection methods

Tab. 1: Comparison of advantages and disadvantages of different methods for site selection

	Advantages	Disadvantages
Iterative Algorithms	Extremely fast Reasonably efficient solutions Easy to implement and comprehensible Offer alternative solutions	A global optimal solution is not guaranteed
Simulated Annealing	More efficient than iterative algorithms Range of reasonably good solutions Objective function with penalties instead of constraints	Time demanding Annealing parameters are set through trial and error A global optimal solution is not guaranteed
Linear Programming	A global optimal solution is guaranteed	Computationally- and time demanding No alternatives are calculated "Black box"-character reduces comprehensibility
Ranking Methods	Fast and comprehensible	Inefficiency Knowledge on single species is lost Complementarity is not taken into account

4.1 Iterative algorithms

The advantages of complementarity-based iterative algorithms can be summarized as follows (see also Pressey et al. 1997, Pressey et al. 1999):

- They produce reasonably efficient solutions,
- they offer alternative solutions (flexibility),

- they can be executed extremely fast and thus allow a comparison of different scenarios even in large planning regions
- they are explicit and produce repeatable results, and
- they may be easily implemented in real-time interactive systems.

Thus, they are especially useful for a rather quick generation of feasible network configurations, to explore different data sets, to test assumptions, and they can be easily implemented for real world planning tasks. Especially, when hundreds of alternative plans incorporating different constraints have to be devised from complex data sets so that these plans can be judged against socio-economic criteria.

The disadvantage of the iterative algorithms refers to the fact that choosing the best site at each step may lead to an only "near-optimal" solution, because the algorithm might get stuck at a "local optimum" (Rebelo & Siegfried 1992, Underhill 1994, Justus & Sarkar 2002, McDonnell et al. 2002). This is first of all caused by a continuous addition of sites to the reserve system without providing an option of removing. Hence, more efficient or less expensive final portfolios will never be discovered when sites are accumulated additively. The potential "value" or contribution of unselected sites to representation targets may be only discovered, if the algorithms are repeatedly applied with different set of sites as starting points. It could therefore be dangerous to assume that the selected sites are necessarily more valuable for nature conservation than the unselected sites (Pressey et al. 1997). Though a "near-optimal" solution does not inevitably might be a disadvantage, at least as long the representation targets are fulfilled. The critical point derives from the question whether the near-optimal solution results in costs that prevent an implementation of the reserve network, whereas a realization of the optimal network solution would have been possible without any restrictions. In order to overcome – at least partly – the inefficiency of iterative algorithms, a check for redundancy should always be part of the algorithm (Pressey & Nicholls 1989a, Williams 1998, Aggarwal et al. 2000).

Site selection algorithms based on identifying hotspots of species richness (richness-based algorithms) tend to be inefficient in maximizing the protection of species diversity, because such hotspots do often not include relatively rare species. In contrast rarity-based algorithms prioritize sites with rare species and hence might reduce the number of sites needed (Reid 1998). However, they might perform not well in choosing reserve networks with many species when the number of sites allowed is small and where sites with rare species do not coincide with species rich sites (Csuti et al. 1997). Often the poorer known and less-well surveyed species, e.g. invertebrates, will have many unique distribution records and many irreplaceable sites (Reyers et al. 2002). Hence, the rarer the species the higher its influence tends to be for site selection (Rodrigues & Gaston 2002). Therefore, a large number of priority sites are selected, if the planning region reveals high local endemism and high species turn-over (Lombard et al. 1999). With respect to the high influence of rare species on site selection it thus should be guaranteed that rarity is "real" and not caused by an inappropriate sampling design.

4.2 Simulated annealing

The trade-off between efficiency and flexibility is solved rather satisfactory with simulated annealing, though a global optimal solution can not be guaranteed (Cabeza & Moilanen 2001). However, the selection process does not get stuck at a local optima, and by repeating the selection process a range of reasonably good solutions to the same problem can be identified.

Disadvantages of simulated annealing result probably from the fact that the annealing parameters are generally set through trial and error. As these parameters can have a large effect on the performance of the selection process this step is rather critical for the results of the selection process. Further, simulated annealing is very time demanding. Finally, iterative algorithms are easier to implement and the selection procedure can be easier explained to the stake-holders and general public involved in the conservation planning process.

4.3 Linear programming

The application of linear programming methods guarantees the identification of optimal solutions by taking into account the given constraints, because all possible site combinations are evaluated (Underhill 1994, Church et al. 1996, Pressey et al. 1996, Camm et al. 1996, Csuti et al. 1997, Önal 2003). Linear programming sometimes may perform only marginally better than iterative algorithms, especially if numerous rare species have to be represented in the reserve network and hence the percentage of irreplaceable sites becomes rather large (Willis et al. 1996, Pressey et al. 1999).

The disadvantage of linear programming is mainly caused by an exponentially increasing number of iterations as a function of the number of planning units (e.g. Cabeza & Moilanen 2001). Thus, it may often take excessive lengths of time to solve conservation planning problems of the size typical for ecoregional planning. In order to reduce processing time Rodrigues et al. (2000a) suggested data reduction, as e.g. the pre-selection of irreplaceable sites or using the linear programming methods in a sub-optimal way by interrupting the program.

According to the aim of identifying the optimal solution no alternative portfolios are identified. Flexibilities in site combination can only be explored if selection criteria, as e.g. spatial components and genetic diversity are added (Önal & Briers 2002, Önal 2003), or if conservations "values" of species and sites are modified (Rodrigues et al. 2000a). Furthermore computational complexity results in a “black box” nature, which reduces comprehensibility for stake-holders or for the general public.

4.4 Ranking methods

A ranking of sites according to single criteria produces fast and comprehensible results (Rossi & Kuitunen 1996). But as complementarity is not taken into consideration these methods fail to account for species turnover between sites and there is no guarantee that priority sites second or third on the list might not just duplicate biodiversity features (Kirkpatrick 1983, Flather et al. 1997, Sarkar & Margules 2002). Thus, ranking methods often do not improve efficiency greatly over ad hoc-representation (Pressey & Nicholls 1989b) and the results rarely coincidence with complementary reserves (Williams et al. 1996a, Csuti et al. 1997, Reid 1998). Notwithstanding it was shown by Lombard et al. (1999) that richness hotspots in areas of high local endemism can incorporate complementarity by default. But they also made the restriction that this result may not be transferable to all planning regions without detailed analyses.

5 Implementation of a systematic site selection process in praxis

5.1 Methods

Successful and effective conservation planning should ideally combine sound biological survey and mapping, statistical modeling, geographic information system analysis, application of efficient site selection methods, experience in on-the-ground implementation, aspects of socioeconomic development, legislation, and political tactics (Pressey & Cowling 2001). Consequently it has to be kept in mind that priority site selection is only one part of the overall biodiversity conservation process. And within this general framework, the question whether methods for place prioritization should be preferred which guarantee global optimal solutions or whether iterative algorithms should be given preference can not be answered with a straightforward yes or no, but probably has to be decided according to the specific situation.

Optimal solutions may be preferred for theoretical exercises or when the size of the data set and the conservation goal are appropriate. Though, for solving the reserve network design problem and even more with respect to the implementation of a reserve network into practice it is often more useful to report multiple solutions that are near optimal than just a single optimal combination of sites (Camm et al. 1996, Prendergast et al. 1999). If one site is unavailable for some reason or if it is destroyed, then other combinations could still meet the overall conservation goal unless it has unique features. Thus, information on flexibilities in network configuration or on the existence of replaceable sites as well as the presentation of a range of good solutions and the implementation of the principle of efficiency will minimize conflicts between land users und facilitate negotiations between interest groups (Pressey et al. 1997, Pressey 1998, Ferrier et al. 2000, Margules et al. 2002). Arthur et al. (1997) even remark that specifying only a single solution would provide incomplete information to the decision makers. In this situation advanced methods such as simulated annealing may provide a good compromise: These methods can deal relatively quickly with complex problems and their solutions are often more efficient than those given by richness- or rarity-based iterative algorithms (Possingham et al. 2000, Cabeza & Moilanen 2001).

Especially when conservation planning projects are carried out in close interaction between planners, regional stake-holders, and decision makers, speed of execution of the selection methods may be an important aspect. Provided that alternative network configurations can be quickly calculated, the consequences of e.g. inclusion or exclusion of planning units, new decisions on land use, changes in economic, social and political conditions, or changes in ecological and biological knowledge on the number, location, and configuration of the priority sites within the planning region can be demonstrated immediately (Bedward et al. 1992, Pressey et al. 1996, Bibby 1998, Lombard et al. 1999, Faith et al. 2001a).

Within this context scenario planning offers an approach for thinking creatively about complex and uncertain futures. The central idea of scenario planning is on the one hand to consider a variety of possible futures that include many of the important uncertainties in the system rather than to focus on the accurate prediction of a single outcome (Peterson et al. 2003). And on the other hand scenario planning should also enable an evaluation of a range of reasonably good solutions in the context of other considerations, such as economics or political expediency (Faith & Walker 2002).

As the process of site selection for nature conservation is not restricted to the scientific community the applied method should be comprehensible for the decision makers and the general public. Managers and conservation practioners who do not understand the algorithms or why a particular place has been identified for conservation will be less supportive of a regional conservation plan than they otherwise might be (Williams et al. 1996b, Bibby 1998, Williams 1998, Theobald et al. 2000, Groves et al. 2002). Thus, the application of repeatable rules and the formulation of goal directed criteria are accepted more readily as a justification for the necessity to set aside networks of sites for nature conservation than an ad hoc-site selection (Justus & Sarkar 2002).

In order to enhance the acceptance of conservation measures the integration of expert knowledge should also be part of the identification of conservation goals, gathering and evaluating data on biodiversity features and threats, or formulation of representation targets. A successful implementation is furthermore supported by a combination of top-down and bottom-up approaches. In this context the top-down approach ensures scientific rigor and the bottom-up approach the representative participation of the relevant stakeholders (Lochner et al. 2003, Younge & Fowkes 2003).

The task of nature conservation does not end with the nomination of priority sites and the dedication of reserves, but instead decisions have to be made on the most appropriate and feasible management measures to ensure the viability of the relevant biodiversity features (Lunney et al. 1997). But management practices may either fail or succeed beyond expectations and species or habitat composition of the sites might evolve in response to environmental and other changes.

5.2 Constraints for an implementation

Most existing reserve selection algorithms assume that a reserve network is designed and sites are selected by decision-makers at a single point in time. In reality, however, selection processes are often dynamic and sites are selected one by one or in several groups because insufficient funds at the beginning of the process restrain the number of sites which could be set aside for conservation. In the mean time, degrading processes might continue, species composition might change or some sites may even be destroyed entirely before they become available for conservation action. As a consequence the final system may be quite poor when measured against the initial goals (Drechsler 2005). Hence, given the ability to protect only a small number of sites a more relevant question might be to ask what proportion of overall biodiversity is captured in priority sites that are identified on the basis of various methodologies (Reid 1998). According to the rapid accumulation of species in relatively few sites, in contrast to the larger number of sites needed for complete representation (Csuti et al. 1997), a "coarse filter" may be applied to represent the major habitat types in relatively few sites (Noss 1996), or to focus initially on sites with a high conservation value (measured e.g. by irreplaceability), but also on sites which are highly vulnerable to processes that threaten biodiversity (Cowling et al. 2003). The conservation needs of species not represented might be addressed on an individual basis ("fine filter").

Despite the numerous advantages of a systematic site selection process it also has to be acknowledged that the site selection process itself might be influenced by various constraints (Pressey & Cowling 2001). While mathematical algorithms for priority site selection continue to improve, it became evident already with the first applications of systematic site selection procedures that the databases for conservation planning have to be improved (Margules et al. 1994, Redford et al. 1997, Spector & Forsyth 1998). Field records are often restricted towards charismatic species or to species that are easy to observe, such as mammals, birds and butterflies, to locations where the species of interest are likely to be found, to easily accessible sites, or towards favorite study sites such as field stations or areas close to major universities or museums (Williams et al. 1996b, Cabeza & Moilanen 2001, Margules et al. 2002, Williams et al. 2002, Pressey 2004). Furthermore, recorded absences are merely available (Freitag et al. 1996). As the quality of site selection depends strongly on the detail, precision, and accuracy of the information being used, further research is needed to understand how sensitive site selection algorithms are to variations in data type, quantity and quality (Possingham et al. 2000).

Summarizing, it should be acknowledged that probably the dichotomy between academic conservation research and applied land use planning will remain and often site prioritization will be governed by non-biological decisions (Prendergast et al. 1999, Scott & Sullivan 2000). Nevertheless, it should be tried throughout the whole selection process to separate questions that can be answered by research based information from questions that can only be answered by policy decisions. In order to strengthen efficient selection procedures, efforts for establishing a sound database are required for nearly all planning regions and furthermore an understanding of ecological and evolutionary processes that maintain and generate biodiversity is required and already existing approaches how to incorporate processes in the selection procedures have to be further developed.

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