WAGENINGEN
UNIVERSITY & RESEARCH

Wettelijke Onderzoekstaken Natuur & Milieu

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WOt-paper 39
December 2016

Assessing biodiversity change in scenario studies

Introducing a decision support tool for analysing the impact of nature policy

Abstract

Biodiversity conservation is firmly established on the political agenda. Nested goals and targets for biodiversity have therefore been formulated and agreed at global, regional, national and sub-national levels in order to halt and reverse its decline. In order to measure progress in relation to the delivery of such targets, policymakers have a range of tools and indicators that allow them to monitor and evaluate the effect of their policies, instruments and associated actions. In terms of the policy cycle, evaluation should result in the further modification and refinement of policy instruments towards improved delivery in the future.

Evaluation can be divided in ex ante and ex post assessments. While for ex post assessments, actual data can be used (i.e. monitoring data, usually combined into indicators), models are necessary for ex ante assessments for forecasting the impacts of policies and measures. Increasingly, such models are being developed in order to provide improved predictive capacity in relation to policy development and implementation. The suitability of such models is determined by more than their scientific merit (credibility); they also need to be trusted by relevant stakeholders (legitimacy) and applied to the needs of decision-makers (saliency) in order to provide effective support to policy processes.

In this paper we introduce the Model for Nature Policy (MNP), developed in order to assess the effects of policies on biodiversity. It assesses biodiversity by stacking results for individual species, that respond to threats and conditions in a species-specific way. The MNP successively models habitat suitability and population persistence, for a set of protected species. It expresses output through policy-relevant indicators. Based on a number of different applications of the MNP its usefulness in the science-policy interface is evaluated in order to provide recommendations for the future development and application of this model and biodiversity models in general. We argue that the applicability of biodiversity models might improve when output is generated in terms of policy-relevant biodiversity indicators and when more attention is paid to the integration of different environmental pressures which policy may influence.

However, experience has shown that striking the balance between scientific credibility, stakeholder legitimacy and saliency for decision makers inevitably leads to a trade-off between the criteria. Identifying such trade-offs and subsequently assessing their impact may help to provide insights into the potential for assessing the effectiveness of such models as policy instruments in the future.

Key words: policy cycle, decision support tool, habitat suitability, biodiversity indicator, species persistence, ecosystem quality

Introduction

Biodiversity is the source of many important goods and services, yet it is declining worldwide. There is consensus that biodiversity loss should be halted, and biodiversity conservation has now become firmly established on the political agenda. At global level the Convention on Biological Diversity (CBD) was signed by 168 government leaders at, and following, the 1992 United Nations Conference on Environment and Development. In 2010 at the tenth meeting of the Conference of the Parties a revised and updated Strategic Plan for Biodiversity, including the Aichi Biodiversity Targets, was adopted for the 2011-2020 period (CBD, 2014). Many policies and measures followed, e.g. the Bird Directive, Habitat Directive, Natura 2000 in Europe.

In 2011, as a response to the CBD's updated Strategic Plan for Biodiversity, the EC adopted a strategy to halt the loss of biodiversity and ecosystem services in the EU by 2020 (EC, 2011). The CBD goals and headline targets are set out in National Biodiversity Strategies and Action Plans (NBSAPs) which are intended to be the principal instruments for implementing the Convention at national level (CBD, 2014). Nested goals and targets for biodiversity have thus been formulated and agreed at global, regional, national and sub-national levels with the aim of halting and reversing its decline, and policies and measures have been adopted at these levels for progress towards achieving these goals and targets. But how can we monitor the effectiveness, or even predict future the impact of policies and measures?

In order to measure progress in relation to the delivery of such targets, policymakers have used a range of indicators that allow them to monitor and evaluate the effect of their policies, instruments and associated actions (Condé *et al*., 2010; Pereira *et al*., 2013). Much progress has been made in the development of biodiversity indicators (Biała *et al*., 2012). However, these indicators are based on data that tracks the historical status and trends in biodiversity and which provide a 'point in time' assessment based on what has gone before (ex post evaluation); they are not specifically intended to evaluate the effects of future policy actions (Leadley *et al*., 2010; Pereira *et al*., 2010; Nicholson *et al*., 2012). As such they rarely provide information on the cause of change or the required policy action (IPBES, 2016).

In terms of the policy cycle, evaluation of progress based on one or more indicators should result in the further modification and refinement of policy instruments towards improved delivery in the future (see for example: Biodiversity Europe DST, 2014). However, in order to achieve this, monitoring data and indicators based upon these data are not enough, as they do not provide sufficient insight into causal mechanisms. Therefore, policymakers stress the need for tools to anticipate and improve the effectiveness of policy instruments for nature conservation and for the sustainable use of biodiversity (EEA, 2012; Hof *et al*., 2015; IPBES, 2016).

Models are increasingly requested and used to provide improved predictive capacity in relation to policy development and implementation in general and specifically in relation to biodiversity (ex ante evaluation, e.g. Alkemade *et al*., 2009; GLOBIO, 2014). The evaluation of conservation policy and the assessment of the potential impact of proposed measures on biodiversity targets also have a spatial component and are therefore becoming important in the context of spatial planning (at a range of scales). Overall, there is an increasing demand for policy-relevant biodiversity indicators and practical tools that are able to establish clear causal relationships between the impact of pressures and these indicators (EEA, 2012).

Biodiversity models can play an important role in this respect, because they can assess the possible impact of potential future measures and policies. However, several studies have indicated the lack of adequate methodologies for accurate, systematic and quantified predictions of impacts on biodiversity (Treweek *et al*., 1993; Thompson *et al*., 1997; Atkinson *et al*., 2000; Byron *et al*., 2000; Geneletti, 2002). Although predictive models have been developed throughout the various ecological disciplines (e.g. Hanski, 1994; Guisan & Zimmermann, 2000; Akçakaya, 2001; Lehmann *et al*., 2002; Scott, 2002; Opdam *et al*., 2003), models that focus on the broad diversity of drivers of biodiversity loss and that address policy-relevant indicators remain scarce.

Different types of biodiversity models can be distinguished (Hof *et al*., 2015). Correlative models relate biodiversity to other factors, e.g., contemporary climatic conditions. Correlative approaches can be further subdivided into models focusing on emergent features of species assemblages (e.g., species richness) and species distribution models (Guisan and Rahbek, 2011). For a biodiversity indicator, individual specific results can be combined (stacked). Mechanistic models explicitly include population dynamics, physiological, or ecological processes affecting species distributions. Again, the output can be a general biodiversity index such as species richness, or species specific results which need to be combined (stacked). Together, the four types of biodiversity model are: correlative assemblage, correlative stacked species, mechanistic assemblage, and mechanistic stacked species. Examples of correlative assemblage models are species richness models (Lemoine *et al*., 2007), and species composition models (Ferrier & Guisan, 2006).

Examples of mechanistic assemblage models are Dynamic global vegetation models (DGVMs, Cramer *et al*., 2001). Examples of stacked correlative models are Species distribution models (SDMs, Thuiller, 2003). Examples of stacked mechanistic models are (semi-) mechanistic SDMs (Engler & Guisan, 2009), and population viability analysis (Lacy, 1993).

Recently, a general ecosystem model has been developed that is applicable at a global scale and for all terrestrial and marine environments (Purves, 2013; Harfoot *et al*., 2014a). This Madingley Model attempts to mechanistically represent whole ecosystems by modelling all the individual heterotrophic organisms in each ecosystem according to their functional traits and based on a set of fundamental ecological processes (Harfoot *et al*., 2014a). Obviously all these categories come with advantages and drawbacks (IPBES, 2016). Mechanistic models tend to be too complicated, too slow, and too data hungry. Statistical models are merely based on correlations, and may lack causality. Stacked models are more complex and time consuming, but assemblage models ignore the fact that species respond differently to changes in conditions. We argue that what is the best biodiversity model in a given situation depends on the question asked and the situation.

Whether models are fit for purpose is not exclusively determined by their scientific credibility (Cash *et al*., 2003); in particular model complexity can seriously hamper practical use and transparency. They should be transparent enough to allow understanding by, and accommodate the wishes of stakeholders. This will increase the likelihood that stakeholders trust the results and view them as reliable (Voinov & Bousquet, 2010; Pouwels *et al*., 2011). Moreover, such models should be able to provide practical answers to policy-relevant questions using a coherent set of indicators (Walpole *et al*., 2009; Sparks *et al*., 2011). An optimal balance between the criteria of scientific credibility, stakeholder legitimacy and salience for decision-makers therefore needs to be established (Cash *et al*., 2003).

The Model for Nature Policy (MNP) is are very simplified mechanistic model. It calculates species specific results and then stacks them into biodiversity proxy indicators such as alpha diversity (Whitakker, 1972). It is based upon mechanistic considerations (population viability analysis) but instead of being a full mechanistic model it uses rules derived from more detailed mechanistic models (Verboom *et al*., 2001; Verboom & Pouwels, 2004) and statistical relations between stress factors and species occurrence. It focusses on major cause–effect relationships for given policy target species. It has already been used in the Netherlands to evaluate national policy plans and to calculate the effects of election manifestos from various political parties; furthermore it has been used in political decision making in the Netherlands. Its development and subsequent use in practice offers an opportunity to explore the issue of striking a balance between scientific credibility, salience for decision-makers and stakeholder legitimacy.

The objective of this paper is to describe the MNP and to set out how it manages to balance the criteria of scientific credibility, salience for decision-makers, and stakeholder legitimacy as formulated by Cash *et al*. (2003). Choices that were made during the design of the MNP in order to establish the scientific credibility and stakeholder trust of the model are discussed in detail. We present three applications to demonstrate the MNP's ability to provide intelligible information to decision-makers by answering three key conservation questions: what is changing; why is it changing; and what can we do about it (UNEP, 2003)? Finally, we discuss the trade-offs that inevitably emerge while looking for a balance and we explore the potential for future improvements.

Model description

General description

The Model for Nature Policy (MNP) consists of three interconnected components that, together, assess the consequences of environmental pressures for biodiversity in general and the delivery of biodiversity policy targets in the Netherlands. The framework shows similarities to that developed by Ferrier & Drielsma (2010) as it also follows three steps: (1) determination of habitat suitability; (2) assessment of expected species persistence; and (3) aggregation of output in order to form policy-relevant indicators such as alpha diversity (Figure 1). The MNP takes into account the impact of three main pressures of desiccation, eutrophication, and fragmentation (Figure 1). These are considered the dominant pressures on biodiversity in the Netherlands and Western Europe (Reijnen *et al*., 2007; Bealey *et al*., 2011; Wamelink *et al*., 2013). The MNP takes into account the processes that influence individual species and aggregates these species-specific responses to form general and policy-relevant biodiversity indicators. The MNP assumes trade-offs between the pressures that are being considered. A reduction in habitat suitability due to desiccation or eutrophication can be compensated for by an increase in habitat area. However, when environmental conditions fall below minimum requirements, the habitat is considered unsuitable, regardless of its size.

The MNP is parameterised for a set of species belonging to the taxonomic groups of vascular plants, butterflies and breeding birds. These three groups were chosen because they operate at various scales of the ecosystem (Carignan & Villard, 2002) and, together, they are a better representation of biodiversity than a single taxon would be (Wolters *et al*., 2006; Eglington *et al*., 2012). In addition, knowledge about habitat preference and environmental sensitivity is available for species within these taxa (Oostermeijer & Van Swaay, 1998; Van Dobben & Van Hinsberg, 2008; Bobbink & Hettelingh, 2011), and they have been selected as focal groups for ecological monitoring in the Netherlands by ecologists and non-governmental organisations (NGOs). Moreover, the selected species are important target species of nature policy both in the Netherlands and in other European countries, and include the species protected under the European Habitats and Birds Directives (EC, 1992; EC, 2009). The species models were calibrated in close cooperation with NGOs and, based on their knowledge of actual species distribution, we selected a set of species consisting of 219 vascular plants, 40 butterflies and 70 breeding birds. The species set contains over three quarters of the vascular plants, butterflies, and breeding birds that are protected under the Habitats and Birds Directives (EC, 1992; EC, 2009).

Determination of habitat suitability

Habitat suitability for individual species at a particular site is considered to be a function of the type of vegetation present, its area and the impact of desiccation and eutrophication on that site (Figure 1). Habitat suitability is defined in terms of population size (Van Horne and Wiens,

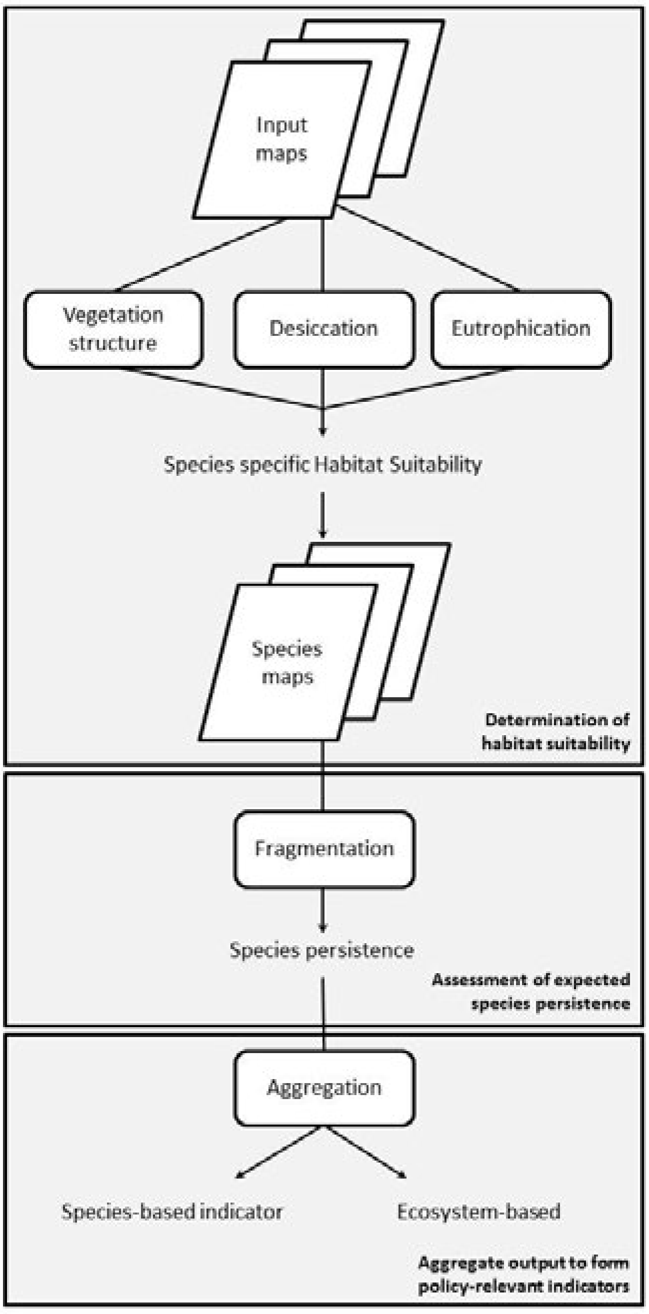

Figure 1 Schematic overview of the three components of the MNP, listing the considered environmental pressures and output indicators. See text for explanation.

1991). Thus: $PS_i = PS_{vt,i} \times A_i \times fE_i \times fD_i$ (Eq. 1)
where i is a grid cell, *PS* is the overall population of a species of site i, $PS_{vt,i}$ is the optimal population density given the type of vegetation at site i, *A* is the size of the grid cell , fE_i is the relative population density based on the state of the driver *E* (Eutrophication), and fD_i is the relative population density based on the state of the driver *D* (Desiccations) of site i.

Information about the vegetation type was derived from monitoring and planning maps used by managers of nature areas. For each type of vegetation, Dutch policy has defined a set of so-called target species. Based on the monitoring data that is collected by a number of Dutch NGOs, the optimal population density for the various target species was calculated per vegetation type. The figures for the population density of individual species were subsequently used to quantify the suitability of habitat areas. The input maps and target species that are used by policymakers, NGOs, and managers of nature areas in the Netherlands were used as input.

For each species, three desiccation levels were assumed: no desiccation; medium; and severe desiccation. Desiccation was defined as a situation where the actual groundwater table would be outside the lowest end of the optimal range for a particular species (Wamelink *et al*., 2013) and the threshold between the states 'none' and 'medium' is set at the level at which population size is reduced to 50% of the optimal population size. The threshold between the states 'medium' and 'severe' is set at the level at which population size is reduced to 10% of the optimal population size. To ensure a close link with water policy and water management, the MNP uses the spring groundwater table to determine the degree of desiccation (Hellegers & Van Ierland, 2003). The MNP model assumes no desiccation when the groundwater table at a particular site is optimal for species occurrence. Optimal groundwater tables for plant species were derived from groundwater tables of the types of vegetation in which the species occurs (Runhaar & Van Walsum, 2004). Optimal groundwater tables for butterflies were based on regression models that relate butterfly occurrence to groundwater conditions (Van Swaay *et al*., 1997; Oostermeijer & Van Swaay, 1998). For breeding birds that are considered to be sensitive to desiccation, optimal groundwater tables were derived from literature (Wilson *et al*., 2004) and by consulting relevant stakeholders at conservation NGOs. The reduction fractions in Equation 1 in situations with no desiccation, medium and serve desiccation are 1, 0.5 and 0.01, respectively. See Table A1 (Appendix 1) for the specific values.

The MNP model defines eutrophication as a situation in which levels of atmospheric nitrogen deposition exceed the critical load. A critical load is defined as 'a quantitative estimate of exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur, according to present knowledge' (Nilsson and Grennfelt, 1988). This measure is commonly used in the abatement of both national and international air pollution (Tuinstra *et al*., 2006), as well as in assessments of environmental impacts on protected areas designated as 'Natura 2000' areas under the EU Habitats and Birds Directives (EC, 1992; EC, 2009; Hicks *et al*., 2011). Again, the model assumes three levels of eutrophication; no eutrophication; medium; and severe levels of eutrophication. No eutrophication is assumed for sites where critical loads are higher than atmospheric deposition. Critical loads for the various plant species were based on empirical (Bobbink & Hettelingh, 2011) and modelled critical loads (Van Dobben & Van Hinsberg, 2008; Van Hinsberg *et al*., 2011) for the types of vegetation that are optimal in relation to those species. For butterflies, the critical loads were based on regression models that relate species occurrence to nitrogen deposition (Oostermeijer & Van Swaay, 1998). For breeding birds that are sensitive to eutrophication, the critical loads of the types of vegetation where these species occur were used. The same information was used to define the deposition levels at which population sizes would reduce to 0.5 and 0.01 of the optimal population size (see Table A2, Appendix 1).

Assessment of expected species persistence

To assess the expected viability of a species within a certain landscape, the output parameter 'species persistence' is determined by using results from habitat suitability modelling as well as an algorithm for the impact of fragmentation (Figure 1). In the model, fragmentation is defined as the situation where single habitat areas are too small and/or too isolated to support a persistent population (Opdam, 1991 ; Ouborg 1993; Fahrig & Merriam, 1994; Tilman & Kareiva 1997). The MNP determines the degree of fragmentation by weighing the size and suitability of habitat areas against the minimum area size and its suitability to accommodate so-called key populations (Verboom *et al*., 2001). Key habitat areas provide species with a survival chance of at least 95% in 100 years within an ecological network such as the National Ecological Network (NEN) (Verboom *et al*., 2001). These areas are crucial for the persistence of species in fragmented landscapes (Opdam *et al*., 2003; Verboom & Pouwels, 2004) and form a source of individual animals that colonise the surrounding areas (Foppen, 2001; Vermaat *et al*., 2008). Area requirements for key habitat areas for butterflies and breeding birds were derived from Verboom *et al*. (2001) and Opdam *et al*. (2008). Area requirements for plant species were based on plant traits related to extinction (Menges, 2000; Reed, 2005; Kleyer *et al*., 2008) and plant densities in the various vegetation types (Schaminee, 2009). Habitat areas that would fall within a species home range are considered to be clustered habitat areas that, together, would meet the area requirements for key habitats (Opdam *et al*., 2003).

Although key habitat areas are important for species persistence, one key area cannot guarantee the long-term survival of a species; a key area does not automatically hold a Minimum Viable Population, but exists as a large habitat area within a wider network (Reijnen *et al*., 2007; Foppen *et al*., 1998). Aiming for a number of large populations will increase the probability of long-term survival (Lindemayer & Lacey, 1995). There is, however, no unequivocal scientific basis to pinpoint the exact number of required habitat areas. Both an upper and a lower threshold were therefore set for each taxonomic group, based on the literature. Above the upper threshold, species are considered to be persistent, whereas below the lower threshold, they are not. The range between both thresholds represents the degree of uncertainty associated with this parameter. For bird species, lower and upper threshold levels were set at 5 and 20 key habitat areas, respectively, based on a study by Foppen *et al*. (1998). For butterflies, higher threshold levels were used, as most of these species are sensitive to environmental fluctuations and therefore require a larger number of habitat areas (Bascompte *et al*., 2002). For them, the lower threshold was set at 20 habitat areas, according to studies by Gurney and Nisbet (1978), Thomas and Hanski (1997), and Baguette (2004). Similar to breeding birds, the upper threshold for butterfly species was set at four times that number. Research into plant species persistence shows that large-scale extinctions are rare for plant species (Honnay *et al*., 2005). As such, persistence thresholds are hard to establish (Menges, 2000). Dormancy of plant seeds and seed banks, periodic recruitment and clonal growth enable plant species to deal with environmental fluctuations (Menges, 2000). For the MNP model, thresholds for vascular plants were therefore assumed to be more likely to be similar to those for bird species than butterflies.

Aggregation of output in order to form policy-relevant indicators

The MNP model aggregates output to form both a species-based and an ecosystem-based biodiversity indicator, linked to relevant policy targets (Figure 1). The species-based indicator provides information on the number of species that, based on habitat suitability and fragmentation assessments, are expected to be viable (i.e. persistent) over the long term, within a certain landscape. In the model, depending on the policy target, sub-selections can be made, such as of national target species (Lammers & Zadelhoff, 1996; Bal *et al*., 2001) or species listed under the Habitats and Birds Directives (EC, 1992; EC, 2009). It is also possible to present results for specific areas in order to provide an estimation of the conservation status of species associated with a particular habitat type or the general conservation status of species present in a certain protected area. This indicator focuses on the species level and is therefore similar to the Convention on Biodiversity (CBD) indicators on species conservation

status and Red List status (EEA, 2007).
The model's ecosystem-based biodiversity indicator assesses the community's richness in species or so-called alpha diversity (Whitakker, 1972) of natural areas by weighing the predicted number of species for a key habitat area, based on the habitat suitability and fragmentation assessments, relative to the total number of species associated with that particular vegetation type (Bal et al., 2001). For each of these types, a threshold number has been defined for 'good' community's richness in Dutch nature policy (Bal *et al*., 2001). This indicator is similar to the CBD's mean species abundance indicator (Alkemade *et al*., 2009), as it weighs the predicted number of species against the potential number of species.

Validation of indicators

General

Modelled species specific output maps were checked by NGOs (SOVON - the Dutch Centre for Field Ornithology, the Vlinderstichting - Dutch Butterfly Conservation). The output of the first step of the MNP, habitat suitability maps, was compared to actual distribution maps of species by experts. Although habitat suitability does not necessarily have to correspond with species occurrence, as both suitable habitat may be (or seem) unoccupied due to local extinction and/or lack of colonization, or merely because the species has not yet been observed/registered at that site, and unsuitable habitat may be (or seem) occupied due to extinction debt (Tilman *et al*., 1996) or observation of non-resident (dispersing) individuals, a positive correlation can be assumed. The output maps of the model were classified into 'good', 'moderate' and bad by the species experts. The expert judgement demonstrated that there is a sufficient match between modelled and empirical data for approximately 69% of the species considered by the MNP. The percentage of good models is the lowest for vascular plants; 59%. In policy assessments we only use those species for which the model output was judged as 'moderate' or 'good'. We also tested the validity of the final indicators by comparing model output against empirical measurements of the present state of biodiversity. In a first comparison, the species-based biodiversity indicator was compared against actual Red Lists of threatened species. The second test compared the spatially explicit output of the ecosystem-based biodiversity indicator against biodiversity hotspot maps in the Netherlands.

Material and methods of model validation using Red list data

The modelled species-based indicator was compared against the current Red List status of the selected species (Van Swaay *et al*., 2010; Bilz *et al*., 2011; BirdLife International, 2015). The Red List status is a widely accepted way of indicating the probabilities of extinction of plant and animal species and is a frequently used indicator in nature policy (EEA, 2007). Criteria such as rate of decline, population size, area of geographical distribution and degree of fragmentation are used to classify Dutch Red List species into 5 groups (Maes & Van Swaay, 1997). These groups are 1) extinct in the Netherlands, 2) critically endangered, 3) endangered, 4) vulnerable, 5) sensitive and 6) non- threatened. It is expected that the 'persistent species' in the MNP model have larger population sizes and larger distribution areas. Therefore they are less affected by fragmentation than non-persistent species and are expected to generally be classified as less threatened according to the Red List.

Results of model validation using Red List data

For the Red list category of non-threatened species, the highest percentage of persistent species was modelled by the MNP, namely 60%. For the category of extinct species this percentage was only 5%. For the in-between categories, the percentage of persistent species was found to gradually reduce from 59% to 47%, to 35% and finally to 21%. Wilcoxon signed-rank tests showed that the distribution of persistent and non-persistent species, according to the MNP over the Red List categories, differed significantly for all three taxonomic groups ($p<0.001$), with more threatened species within the non-persistent set.

Material and methods of model validation using distribution data

The modelled ecosystem-based biodiversity indicator (Figure 3) was compared to field data. National maps that indicate how many target species of certain habitat types are present – so-called hot-spot maps – were derived from nation-wide survey data provided by NGOs. Similarly to MNP output, they also were based on target species within the taxonomic groups of vascular plants, butterflies and breeding birds (Van Hinsberg *et al*., 2011). Habitat areas with modelled high values for ecosystem-based biodiversity indicators are generally expected to accommodate a larger number of species.

Results from model validation using distribution data

Pearson's Product-Moment Correlation confirms the expected positive correlation coefficients, which were found for 11 of the 12 ecosystems considered by the MNP. However, correlation coefficients were rather low and ranged from 0.2 to 0.66 (all significant at $p<0.01$), showing that hot spots based on occurrence could only be partially predicted by modelled 'community's richness in species' values.

Case studies

Three different recent applications of the MNP model are discussed below in order to illustrate the model's ability to contribute to the process of decision-making in relation to policy.

Case 1: Evaluating the conservation status of biodiversity in The Netherlands

Between 2005 and the present the MNP model has been used bi-annually to assess the status of the conservation of species and habitats in relation to policy targets. (PBL, 2010; PBL, 2012).

For these status analyses, the MNP model was used to evaluate the configuration of the National Ecological Network (NEN), as well as the impacts of selected environmental pressures. Both species-based and ecosystem-based biodiversity indicators were used to provide information on the conservation status of species and habitats protected under national law or the EU Habitats and Birds Directives (EC, 1992; EC, 2009). The results show the 'gap' that needs to be bridged in order to meet the biodiversity targets set by the Dutch Government, the CBD, and the European Commission (Figure 2). MNP model output can also be presented in detailed maps (Figure 3), which show the ecosystem-based biodiversity indicator 'community's richness in species' within the present extent of the NEN. Model results can also be used to assess the impact of pressures that drive the ongoing loss of biodiversity and to show why policy targets are not being met. Such a link between state indicators and pressure indicators is often lacking in indicator sets (EEA, 2012; Walpole *et al*., 2009). The MNP's assessment of the NEN in 2012 predicted 58% of the target species to be threatened due to at least one environmental pressure. In addition, more than half of these species were shown to be affected by multiple pressures. In total, 17% of the species lacked suitable habitat; for another 17% the available habitat area was too fragmented to ensure persistence. Desiccation and eutrophication were shown to cause the number of persistent species to be diminished by another 14% and 10%, respectively.

Case 2: Assessing the impact of proposed measures

The model can be used to calculate the effects of proposed, new or existing policy measures: ex-ante evaluation. As such, the model was used to calculate the effects of the 2012 election manifestos in the Netherlands. Among others, the model was used to inform policy-makers about the ecological consequences of budget cuts in biodiversity policies. In the aftermath of the 2008–2009 financial crisis, the Dutch Government planned large budget cuts and lowered its ambition levels regarding the realisation of the National Ecological Network. Because the proposed measures had not yet been made explicit, a range of scenarios were explored. Figure 4 presents and explains model results showing the effect of different measures on species persistence in a cumulative stack.

Case 3: Interactive scenario evaluation; assessing the impact of alternative configurations of nature

To facilitate ongoing discussion about the future of nature policy, the MNP was applied in interactive scenario evaluation (Pressey *et al*., 2009), exploring alternative spatial configurations and conservation motives for nature areas in order to provide insight into what could be done in the future. These scenarios (PBL, 2012) were explicitly framed as possible, not probable, futures (Peterson *et al*., 2003) in order to discover the relationship between biodiversity conservation and other motives for nature conservation. As such, they inform the political debate beyond the context of already adopted Dutch nature policy targets.

Dominant motives for nature conservation were identified using a scenario method (Dammers & Evers, 2008). Relevant stakeholders, including policy-makers, were consulted during a series of participative workshops (Dammers *et al*., 2011) to warrant legitimacy. Four motives were identified: (1) Biodiversity conservation; (2) ensuring the increased and sustainable use of regulating

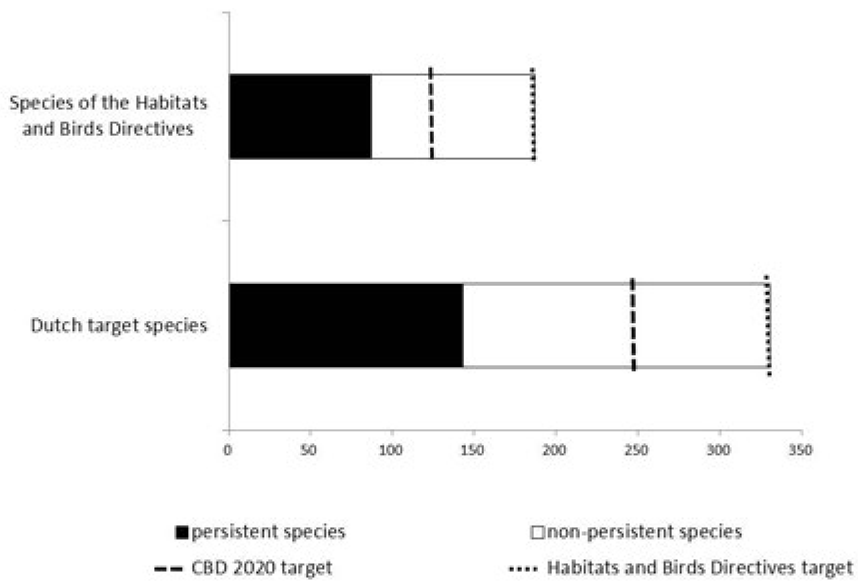

Figure 2 Evaluation of the state of biodiversity in the Netherlands in 2012 and the distance to policy target levels. 'Persistent species' relates to the species-based biodiversity indicator, i.e. the number of species predicted to meet the MNP upper threshold for species persistence.

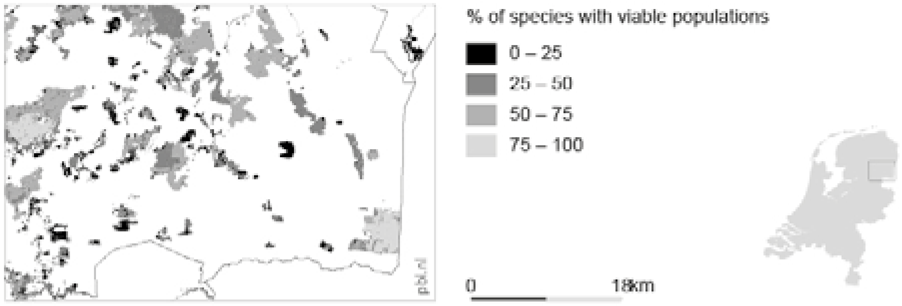

Figure 3 Map of the ecosystem-based biodiversity indicator, based on species meeting the MNP key habitat criteria.

Ecosystem services; (3) enhancing the potential of nature for Recreation; and (4) providing more room for the Economy by allowing developments in and around nature (Van Oostenbrugge *et al*., 2010). For these four motives a future scenario for the Netherlands was described using storylines. These storylines were elaborated spatially explicit by locating parts of the Netherlands where it is expected that the different motives will lead to changes (Dammers *et al*., 2011). These changes were quantified and he MNP was used to assess the biodiversity impacts resulting from these scenarios.

Comparing the MNP results generated by the contrasting scenarios provides insight into the impact of targeting other motives for conservation (Sijtsma *et al*., 2011). Interrelations between motives become apparent (Figure 5). There are, for instance, seemingly strong synergies between biodiversity conservation and the provision of regulating ecosystem services. Furthermore, the results show that creating nature areas with a high recreational potential does not automatically result in high biodiversity value.

Discussion and Conclusions

The objective of this paper is to describe the MNP and to set out how it manages to balance the criteria of scientific credibility, salience for decision-makers, and stakeholder legitimacy as formulated by Cash *et al*. (2003). Before we revisit these criteria one by one, we discuss some of the issues raised in the case study section.

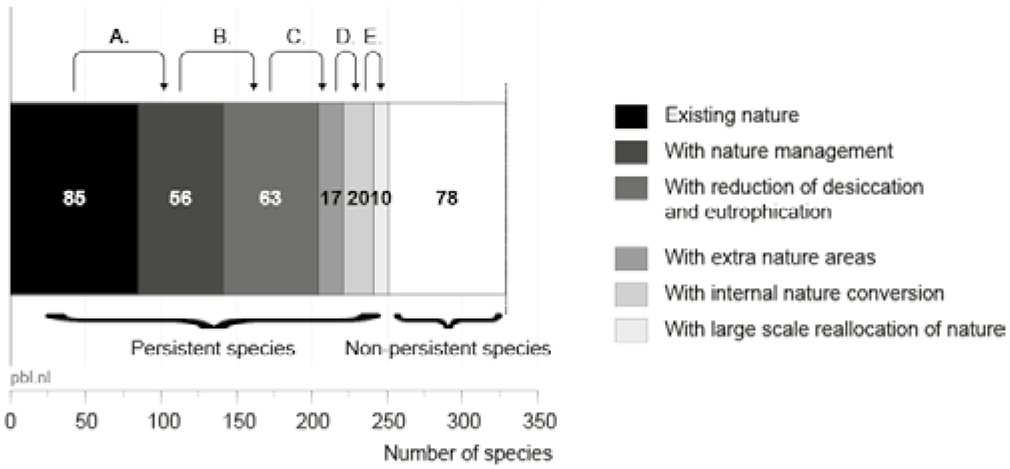

Figure 4 MNP model results: A ranked and cumulative stack of the ecological impact of measures on species persistence (the species-based biodiversity indicator) in the National Ecological Network (NEN). The existing natural areas would be sufficient for 85 persistent species. Additional budgets for various management actions would result in an increase of persistent species. The evaluated actions are: additional budgets for (A) nature management e.g. to mitigate the impact of eutrophication (such as grazing in semi-natural grassland or sodcutting of heathlands) ; (B) reducing the causes of desiccation and eutrophication (such as reducing intensive agricultural practices near large nature areas) ; (C) enlarging nature areas; (D) conversion of common ecosystems, like dry woodlands, into rare ecosystems, like heathlands; and (E) large-scale reconfiguration of nature areas.

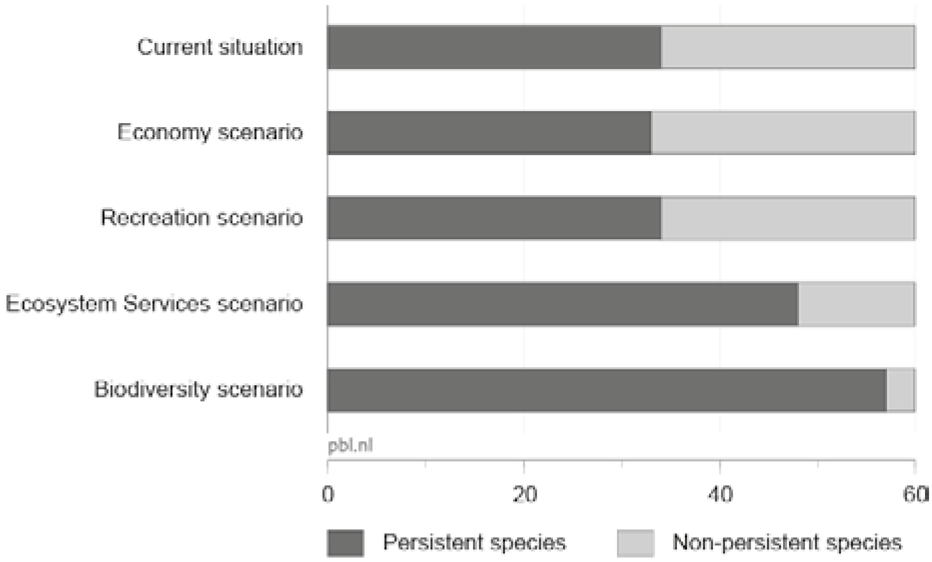

Figure 5 The MNP species-based biodiversity indicator for the current situation and for four contrasting scenarios.

1 The species-based biodiversity indicator seems to be most sensitive for policies that aim at increasing the size and improving the quality of already large nature areas. This is in line with recommendations for efficient conservation (Hodgson *et al*., 2011) and ecological networks in the Netherlands (Ovaskainen, 2013) and England (Lawton *et al*., 2010).
2 Individuals of particular species may be present at sites that according to the MNP model would be considered unsuitable for long-term viable populations; That can point at a flaw of the model or input datasets (vegetation, eutrophication, desiccation), or point at a real mismatch between current circumstances and species distribution, for instance, due to extinction debt in small and/or degraded areas (Tilman *et al*., 1994).
3 Maps such as Figure 3 can be a valuable communication means in stakeholder dialogue. They can be used when engaging local stakeholders in decision-making processes, as they show the spatial pattern of challenges and opportunities (Pouwels *et al*., 2011). Moreover, information about identified bottlenecks provides policy-makers with leads on what they could do to ensure that biodiversity targets will be achieved (Wamelink *et al*., 2013).
4 Both the species-based and the ecosystem-based biodiversity indicators may provide decision-makers with information when considering options to achieve biodiversity targets that have been set. The results shown in Figure 4 were incorporated into a set of policy measures that could form an alternative to existing policy (PBL, 2011), which served as input for the development and assessment of a new administrative agreement on nature policy (EZ, 2011).
5 Since these types of results bring together stakeholders who all have different objectives, identifying overlapping and conflicting interests and creating a shared understanding of future challenges, as a consequence, not only the MNP but also the scenarios themselves can be considered as boundary objects (Star & Griesemer, 1989) that bridge the gap between scientists, relevant stakeholders, and decision makers (Nicholson *et al*., 2012).

Credibility

MNP was built on information derived from the literature, other models and experts. We have argued that model output is statistically correlated with species distribution data, Red List status as well as hot-spot maps. However, correlations are not that high and our choices to improve salience might have had a negative impact on credibility. Our decision to focus on the impact of dominant environmental pressures on biodiversity may have improved the MNP's policy relevance, as targets are expressed in policy terms. However, such relationships are often less straightforward than is assumed in the model. Plants, for example, do not respond to changes in average groundwater levels, but to associated temporary water stress and oxygen shortages (Bartholomeus *et al*., 2011). Something similar holds for eutrophication: Species occurrence negatively correlates with nitrogen deposition (Van Hinsberg *et al*., 2011), but the underlying causal relationship is driven by toxic ammonia concentrations in the air (Sutton *et al*., 2011), critical changes in soil chemistry (Bobbink & Hettelingh, 2011) and shifts in competition between species (Tamm, 1991; Aerts & Chapin, 2000; Bobbink *et al*., 2003). Sensitivity analysis could prove to be valuable in guiding future improvements in the model and identify highly uncertain parameters that exert a great influence on the indicator values.

Legitimacy
We increased legitimacy by consulting relevant stakeholders at various moments in time, allowing trust to be built. Vegetation and land-use maps from NGOs and governmental organisations were used as model input (Reijnen *et al*., 2007). NGOs were also involved in discussions about the sensitivity of target species to the environmental pressures that were assessed. Moreover, the taxa considered by the MNP model also serve as target groups in ecological monitoring programs carried out by the stakeholders. In addition, the applied ecosystem quality indicator presents output in detailed maps, allowing stakeholders to use the model output, identify challenges and contribute their knowledge (Pouwels *et al*., 2011). It can be argued that the degree of stakeholder legitimacy was restricted by prioritising the model's policy relevance. Dutch nature policy targets primarily relate to ecological targets. However, species and ecosystems may also have other value to stakeholders, such as through the provision of ecosystem services, economic benefits, or recreational potential (TEEB, 2010). Although these other alternative conservation motives are given more attention in current policy no explicit targets are set yet. To ensure continued legitimacy the Dutch nature policy as well as the MNP model should assimilate these.

Salience
Enhancing the salience of the MNP model to decision-makers had a high priority during model design. Indicators are aligned with policy targets and the pressures that are considered are targeted by environmental legislation. The MNP model assesses the impact of these pressures on biodiversity in a DPSIR-based framework (Drivers, Pressures, States, Impacts, Response) (EEA, 1999). As such, the model is able to provide decision-makers with a storyline that tells them which policies will be effective for protecting biodiversity (UNEP, 2003). The benefits of this approach in relation to salience levels is demonstrated by the fact that the MNP model played an important role in supporting the key decision moments indicated in this paper. Since the time of the presented case studies, Dutch nature policy has shifted priorities away from the NEN and the so-called robust corridors towards the conservation of species and habitats listed by the EU Habitats and Birds Directives (EC, 1992; EC, 2009). In order to remain salient, the MNP model has to accommodate not only advancing scientific insights but also changing perceptions. Such adaptions can be implemented with relative ease, as the MNP model was built using a flexible modular architecture (Maxwell & Constanza, 1997; Scheller *et al*., 2007).

There is no general approach to ensure an optimal mix between scientific credibility, salience for decision-makers and stakeholder legitimacy (Cash *et al*., 2003). However, we argue that the establishment of straightforward dose-response algorithms, causal links between pressures and their impact on biodiversity indicators, trust by stakeholders, and a set of understandable policy-relevant indicators related to given policy targets are all important aspects. As these success factors were all taken into account during model design, the MNP model is able to bridge the gap between scientists, relevant stakeholders and decision-makers. As such, it acts as a common platform for discussion and collaboration related to Dutch nature policy. It facilitates trans-boundary collaboration (Huitema & Turnhout, 2009) by being objective and accountable to all stakeholders involved. Trade-offs between the criteria of credibility, salience and legitimacy listed by Cash *et al*. (2003) require constant attention in order for the MNP model to remain fit for purpose. Identifying those trade-offs and subsequently assessing their impact provides a perspective from which to enhance the effectiveness of the model at the science-policy interface in the future.

The Netherlands has a long tradition in applying biodiversity models at the science-policy interface (Kros *et al*., 1995; Alkemade *et al*., 1998). In the past the usefulness of model-chains was determined by their scientific quality in describing biodiversity. The models were therefore largely determined by scientists and professional ecologists rather than stakeholder groups representing policy makers and those who might act on the policy. However, in accordance with the theory of Chwif *et al*. (2000), as the modelling increases in complexity this leads to restricted use of the models in policy use (Vader *et al*., 2004). In contrast to previous models, the MNP presented in this paper (a) simplifies complex dynamic processes to simple dose-response relationships, (b) aims to build trust by accommodating stakeholder wishes (Voinov & Bousquet, 2010; Pouwels *et al*., 2011) and (c) communicates output through straightforward policy-relevant indicators (Nicholson et al., 2012). We believe that the practical applications of the model presented here indicate that this approach has enabled the MNP model to contribute to important moments in decision-making in relation to Dutch nature policy by answering three key conservation questions: what is changing, why is it changing, and what can we do about it (UNEP, 2003)? We believe that the MNP model's relevance at the Dutch science-policy interface can be explained largely by its ability to balance scientific credibility with stakeholder legitimacy and salience for policymakers (Cash *et al*., 2003). However, striking an optimal balance between these criteria means that the trade-offs between them have to be made explicit.

Acknowledgements
This research has been funded since 2005 by the Dutch Ministry of Economic Affairs and was commissioned by the Statutory Research Tasks unit for Nature and the Environment, part of Wageningen University and Research. The authors thank Harold Kuipers and Michiel van Eupen for their technical support during the development and use of the model. Unfortunately the developer of the model, Harold Kuipers, passed away in 2008. Finally the authors thank Jaap Wiertz and Paul Opdam for their thoughtful comments on the conceptual model and the paper. The final version of the paper was reviewed by Wieger Wamelink.

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Appendix 1 Example of parameters for nine species

For nine species (three plant species, three butterflies and three bird species) parameters are given as an example. Plant species and butterflies are more sensitive to the impact of eutrophication (parameters for Ndep; Table A1) and birds are more sensitive to fragmentation (parameters for key area; Table A2).

Table A1 Parameters for Bog Asphodel, Meadow Thistle, Marsh Cinquefoil, Silver-studded Blue, Large Heath, Ilex Hairstreak, Nuthatch, Bittern, Sky Lark. Thresholds regarding the impact of Nitrogen deposition (Ndep) are given in mol N/ha/jr and of desiccation (GVG) in cm below the ground surface. Carrying capacity is given as a ratio compared to optimal habitat (=1) (CC_factor) for each ecotope (LS_type), the impact of desiccation is given by four parameters (GVG_L20, GVG_L80, GVG_H80 en GVG_H20; see also the factors a1, b1, b2 en a2 in Figure A1) and eutrophication is given by four parameters too (Ndep_L20, Ndep_L80, Ndep_H80 en Ndep_H20.

Species name	LS_type	CC_ factor	GVG_ L20	GVG_ L80	GVG_ H80	GVG_ H20	Ndep_ L20	Ndep_ L80	Ndep_ H80	Ndep_ H20	Groep
Bog Asphodel	hz-3.10	1	-5	5	43	54	0	0	1190	1750	plant
Bog Asphodel	hz-3.7	0,006	-5	5	43	54	0	0	1190	1750	plant
Marsh Cinquefoil	az-3.4	0,018	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	du-3.4	0,041	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	du-3.5	0,603	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	du-3.9	0,572	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	hz-3.3	0,486	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	hz-3.4	0,622	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	hz-3.7	1	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	lv-3.3	0,426	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	lv-3.4	0,523	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	ri-3.3	0,173	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	ri-3.4	0,125	-46	-29	2	14	0	0	1560	2240	plant
Marsh Cinquefoil	zk-3.4	0,024	-46	-29	2	14	0	0	1560	2240	plant
Meadow Thistle	du-3.5	1	-6	3	30	39	0	0	760	1390	plant
Meadow Thistle	du-3.9	0,239	-6	3	30	39	0	0	760	1390	plant
Meadow Thistle	hz-3.10	0,234	-6	3	30	39	0	0	760	1390	plant
Meadow Thistle	hz-3.7	0,772	-6	3	30	39	0	0	760	1390	plant
Meadow Thistle	lv-3.4	0,621	-6	3	30	39	0	0	760	1390	plant
Meadow Thistle	ri-3.4	0,003	-6	3	30	39	0	0	760	1390	plant
Ilex Hairstreak	du-3.10	0,25	80	80	99999	99999	350	530	700	2810	vlinder
Ilex Hairstreak	du-3.11	0,333	80	80	99999	99999	350	530	700	2810	vlinder
Ilex Hairstreak	du-3.12	0,5	80	80	99999	99999	350	530	700	2810	vlinder
Ilex Hairstreak	du-3.13	0,5	80	80	99999	99999	350	530	700	2810	vlinder
Ilex Hairstreak	hl-3.8	0,8	80	80	99999	99999	350	530	700	2810	vlinder
Ilex Hairstreak	hz-3.11	0,5	80	80	99999	99999	350	530	700	2810	vlinder
Ilex Hairstreak	hz-3.12	0,333	80	80	99999	99999	350	530	700	2810	vlinder
Ilex Hairstreak	hz-3.13	0,25	80	80	99999	99999	350	530	700	2810	vlinder
Ilex Hairstreak	hz-3.9	0,25	80	80	99999	99999	350	530	700	2810	vlinder
Large Heath	hz-3.10	1	-16	-11	17	25	350	380	420	2000	vlinder
Large Heath	lv-3.6	0,5	-16	-11	17	25	350	380	420	2000	vlinder
Silver-studded Blue	du-3.8	0,5	-99999	-99999	99999	99999	350	450	1500	2000	vlinder
Silver-studded Blue	du-3.9	0,050	-12	-7	69	99999	350	450	1500	2000	vlinder
Silver-studded Blue	hl-3.4	0,5	70	90	99999	99999	350	450	1500	2000	vlinder
Silver-studded Blue	hz-3.10	0,9	-12	-7	69	99999	350	450	1500	2000	vlinder
Silver-studded Blue	hz-3.7	0,167	-12	-7	69	99999	350	450	1500	2000	vlinder
Silver-studded Blue	hz-3.8	0,5	-99999	-99999	99999	99999	350	450	1500	2000	vlinder
Silver-studded Blue	hz-3.9	0,5	-99999	-99999	99999	99999	350	450	1500	2000	vlinder
Bittern	az-3.4	0,5	-99999	5	25	25	-99999	-99999	99999	99999	vogel
Bittern	du-3.4	0,5	-99999	5	25	25	-99999	-99999	99999	99999	vogel
Bittern	hl-3.3	0,5	-99999	5	25	25	-99999	-99999	99999	99999	vogel
Bittern	hz-3.3	0,5	-99999	5	25	25	-99999	-99999	99999	99999	vogel
Bittern	lv-3.3	0,5	-99999	5	25	25	-99999	-99999	99999	99999	vogel
Bittern	ri-3.3	0,5	-99999	5	25	25	-99999	-99999	99999	99999	vogel
Bittern	zk-3.4	0,5	-99999	5	25	25	-99999	-99999	99999	99999	vogel
Nuthatch	az-3.7	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	az-3.8	0,25	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	du-3.12	0,25	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	du-3.13	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	du-3.14	0,625	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	du-3.16	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	hl-3.10	0,75	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	hl-3.11	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel

Species name	LS_type	CC_ factor	GVG_ L20	GVG_ L80	GVG_ H80	GVG_ H20	Ndep_ L20	Ndep_ L80	Ndep_ H80	Ndep_ H20	Groep
Nuthatch	hz-3.13	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	hz-3.14	0,75	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	hz-3.15	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	hz-3.18	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	hz-3.19	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	ri-3.10	0,167	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	ri-3.12	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	ri-3.9	0,75	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	zk-3.10	0,25	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Nuthatch	zk-3.13	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	az-3.1	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	az-3.2	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	az-3.3	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	az-3.5	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	az-4.1	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	du-3.3	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	du-3.5	0,25	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	du-3.6	1	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	du-3.7	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	du-3.8	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	du-3.9	0,25	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	du-4.1	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	du-4.2	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	gg-3.1	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	gg-3.2	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	hl-3.4	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hl-3.5	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hl-3.6	0,25	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hl-3.7	0,25	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hl-4.1	0,75	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	hl-4.2	0,25	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	hz-3.10	0,45	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hz-3.5	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hz-3.6	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hz-3.7	0,333	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hz-3.8	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hz-3.9	1	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	hz-4.1	0,75	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	hz-4.2	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	lv-3.4	0,65	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	lv-3.5	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	lv-4.1	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	lv-4.2	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	ri-3.4	0,4	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	ri-3.5	0,5	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	ri-3.6	0,25	-99999	-99999	99999	99999	350	350	890	2000	vogel
Sky Lark	ri-4.1	0,75	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	ri-4.2	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	zk-3.3	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	zk-3.5	0,75	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	zk-3.6	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	zk-4.1	1	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel
Sky Lark	zk-4.2	0,5	-99999	-99999	99999	99999	-99999	-99999	99999	99999	vogel

Table A2 Parameters for nine species. 'Key_area' is the threshold of optimal habitat for a key patch in ha's., 'loc_dist' is the distance to cluster patches as local populations in meters, 'CC-factor' the impact of climate change on the suitability of habitat, 'CCKey_factor' the impact of climate change on the size of the key patch due to weather extremes and # keys the threshold of key patches needed for a species to be viable.

Name	key_area	locdist	CC_factor	CCKey_factor	# keys
Bog Asphodel	50	100	0.5	1.2	20
Meadow Thistle	50	100	2	1.5	20
Marsh Cinquefoil	50	100	1	1.5	20
Silver-studded Blue	5	50	1	3	80
Large Heath	5	50	0.25	3	80
Ilex Hairstreak	50	50	1	3	80
Nuthatch	50	100	1	1.5	20
Bittern	300	300	0.5	1.2	20
Sky Lark	300	100	1	2	20

Colofon

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ISSN 1879-4688

Research conducted by WOT Natuur & Milieu is financed by the Ministry of Economic Affairs.

This paper was produced in accordance with the Quality Manual of the Statutory Research Tasks Unit for Nature & the Environment

Project WOt-04-011-036.61
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