

Modeling direct and indirect climate change impacts on ecological networks: a case study on breeding habitat of Dutch meadow birds

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Received: 1 April 2014 / Accepted: 13 December 2014 / Published online: 6 January 2015
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Abstract

Context Climate change can directly affect habitats within ecological networks, but may also have indirect effects on network quality by inducing land use change. The relative impact of indirect effects of climate change on the quality of ecological networks currently remains largely unknown.

Objectives The objective of this study was to determine the relative impact of direct and indirect effects of climate change on a network of breeding

habitat of four meadow bird species (Black-tailed godwit, Common redshank, Eurasian oystercatcher and Northern lapwing) in the Netherlands.

Methods Habitat models were developed that link meadow bird breeding densities to three habitat characteristics that are sensitive to environmental change (landscape openness, land use and groundwater level). These models were used to assess the impact of scenarios of landscape change with and without climate change on meadow bird breeding habitat quality for a case study area in the peat meadow district of the Netherlands.

Results All scenarios led to significantly reduced habitat quality for all species, mainly as a result of conversion of grassland to bioenergy crops, which reduces landscape openness. Direct effects of climate change on habitat quality were largely absent, indicating that especially human adaptation to climate change rather than direct effects of climate change was decisive for the degradation of ecological network quality for breeding meadow birds.

Conclusions We conclude that scenario studies exploring impacts of climate change on ecological networks should incorporate both land use change resulting from human responses to climate change and direct effects of climate change on landscapes.

Electronic supplementary material The online version of this article (doi:10.1007/s10980-014-0140-x) contains supplementary material, which is available to authorized users.

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Keywords Black-tailed godwit · Common redshank · Eurasian oystercatcher · Northern lapwing · Bioenergy crops · Land use change

Introduction

Climate change is considered as one of the major threats to biodiversity and species conservation (Parmesan et al. 2013). Biogeographical models predict latitudinal and altitudinal shifts of species ranges in response to increased temperatures, which are corroborated by empirical studies (Parmesan and Yohe 2003; Burrows et al. 2011). In addition to effects on the physiology of species, negative climate change impacts are related to reductions in habitat quality, e.g. resulting from lower groundwater levels and increased amplitudes of soil moisture conditions (Mustin et al. 2007; Van Teeffelen et al. 2014; Witte et al. 2014), as well as to the increased frequency of extreme weather events, which directly affect population size (Parmesan et al. 2000; Verboom et al. 2010; Van Teeffelen et al. 2014). Climate change can be especially detrimental to populations of species that depend on ecological networks of habitat patches within a hostile matrix (e.g. urbanized or agricultural areas). The impact of climate change on such species strongly depends on the quality of the network, both in terms of connectivity between patches and the susceptibility of those patches to environmental change (Vos et al. 2008; Verboom et al. 2010).

Apart from direct climate change effects on habitat quality, indirect effects in intensively used landscapes may result from land use change as a human adaptation to climate induced reductions of agricultural yields and overall profitability (Opdam et al. 2009; Mandryk et al. 2012; Kanellopoulos et al. 2014; Bakker et al. 2014). Such land use change may have considerable negative effects on both the connectivity and the habitat quality within individual patches of ecological networks. A common adaptation option in agriculture that is explored is growing bioenergy crops. A large scale switch to bioenergy crops may lead to significant landscape changes (Gaucherel et al. 2010). Although it is increasingly recognized that projections of climate change impacts on ecological network quality also should include effects of autonomous land use changes (Barbet-Massin et al. 2012; Van Teeffelen et al. 2012; Jongsomjit et al. 2013; Riordan and Rundel 2014), land use changes driven by climate change itself are rarely taken into account (but see Bakker et al. 2014). The relative impact of such indirect effects of climate change on the quality of

ecological networks therefore remains largely unknown.

In this paper, we analyze direct and indirect effects of climate change on habitat quality, using breeding meadow birds in the peat meadow landscape in the Netherlands as an illustrative case study. The peat meadows in the Western part of The Netherlands are currently one of the strongholds for breeding meadow birds in Europe. An estimated 40 % of the European population of Black-tailed godwit (*Limosa l. limosa*) breeds in The Netherlands (BirdLife International 2004). Due to increased agricultural intensification and other land use change, populations of meadow birds are in strong decline. Black-tailed godwit for instance is now categorized as a near-threatened species on the IUCN Red List (BirdLife International 2014). Meadow birds depend on a network consisting of larger, contiguous peat meadow areas within the highly urbanized western part of the Netherlands for breeding and chick rearing (Melman et al. 2008; Van der Vliet et al. 2014), whereas pre- and post-breeding staging areas are largely located in tidal flats and river floodplains (Van der Vliet et al. 2014). Parts of the network of staging areas have been appointed as Special Protection Areas under the EU Bird Directive and are part of the Dutch Natura 2000 network. The core breeding areas, however, remain unprotected farmland. Peat meadows are considered specifically vulnerable to the effects of climate change (Querner et al. 2012; Van Bodegom et al. 2014). If climate-induced land use change leads to reduction of habitat quality in these core breeding areas, this indirect effect of climate change could potentially accelerate population decline.

To assess the impact of changes in peat meadow landscapes on meadow bird densities in response to direct and indirect effects of climate change, we first developed a series of simple habitat quality models that link meadow bird densities to groundwater level, land use and landscape openness, which were parameterized with data from a nationwide census. Second, we developed two landscape change scenarios for a case area in the Dutch peat meadow district, one with and one without climate change. Finally, the habitat models were used to estimate future meadow bird densities in the case area as a response to the changes in land use, groundwater level and landscape openness in each scenario.

Methods

Bird data

We developed habitat models for four species of meadow birds: Black-tailed godwit, Common redshank (*Tringa totanus*), Eurasian oystercatcher (*Haematopus ostralegus*) and Northern lapwing (*Vanellus vanellus*), which differ in their breeding ecology. During the breeding season, these species are easy to survey. For the Netherlands as a whole, relative density maps are available (Sovon 2002). Relative density is defined as the probability of recording a species within a certain period. Input data for these maps were collected during the breeding seasons between 1998 and 2000 on the level of a continuous grid of 5 × 5-km squares, consisting of 25 1 × 1-km grid squares. Eight of these 25 1 × 1-km grid squares were situated in a fixed pattern and were surveyed twice during the breeding season for all behavior indicative of breeding within 1 h of observation (Van Turnhout and Vogel 1997). Surveys were carried out by volunteers using a standardized protocol. Relative density for all non-surveyed 1-km squares was calculated by kriging, using field data from the 12 nearest surveyed 1 × 1-km squares in the fixed grid with the same land use (Sovon 2002). In most cases those 12 squares were within a distance of 4 km, although the upper limit for this kriging distance was set at 25 km. We downscaled relative density maps to a grid of 500 × 500 m squares to match the grid size of the groundwater level map that was available for this period. Van Turnhout and Vogel (1997) and Sovon (2002) provide more details on sampling and interpolating methods.

Habitat suitability models

Based on their importance to meadow bird habitat quality and their expected sensitivity to climate and land use change, we selected groundwater level, land use and openness as main habitat variables in our regression models. We used the average spring groundwater level (ASG), which is defined as the annual average of three measurements of groundwater level (below the soil surface) in the period 14 March–14 April for a period of 8 consecutive years (cf. Runhaar et al. 1997), rounded to the nearest centimeter. Among several measures for groundwater level,

ASG is the most relevant for our study because meadow bird species arrive and settle territories in March–April. Groundwater data were not available for urban areas and the hilly extreme south of the Netherlands. Meadow birds generally do not breed in these areas and our models were only applied in the flat western part of the country. Also, meadow birds are almost absent from urban areas, whereas the groundwater level there is held constant to protect these areas against the effects of soil subsidence. We therefore assumed that this lack of groundwater data had little influence on our model outcomes. Groundwater data had a resolution of 25 × 25 m, which was resampled to 500 × 500 m using the average of all 25 × 25 m cells contained within each 500 × 500 m grid cell of the output map.

We distinguished 11 categories of land use based on the Dutch land-cover database (LGN) from 1997 (Thunnissen and De Wit 2000). A list of all land use categories is given in Appendix 1. The original land cover map had a resolution of 25 × 25 m, which was resampled to 500 × 500 m, assigning the value of the most abundant land use class within each cell.

For openness, we used a nationwide map with a grid of 1-km squares in which openness was classified from 1 (very open) to 9 (very closed) based on area percentages of built up area and vegetation >2 m (Dijkstra and van Lith-Kranendonk 2000; Appendix 2). This map was resampled to 500 × 500 m grid size.

We used an information-theoretic (IT) approach (Burnham and Anderson 2002) to analyze relationships between bird density and the three landscape characteristics. The IT approach conducts a comparative model fit for a group of competing models, thereby avoiding the drawbacks of automated stepwise algorithms (Whittingham et al. 2006). This method also avoids problems of spatial autocorrelation in regression (i.e., overestimating significance by standard significance tests in parametric regression; Hawkins 2012). We used the generalized linear models (GLMs) procedure with a Poisson error distribution and a log-link function in PASW Statistics 18 (SPSS Inc. 2009) to fit all models based on all possible combinations of habitat variables. Interaction terms were not considered. We used a (log) linear link function because preliminary screening of the data did not suggest curvilinear relationships between the explanatory variables and bird density data. Collinearity diagnostics showed that the analysis was not

influenced by collinearity of the explanatory variables (tolerance > 0.1 and VIF < 10; cf. Field 2009).

Akaike weights (w_i) calculated from the Akaike information criterion (AIC) were used to obtain information about the relative fit of each model to the data (Burnham and Anderson 2002; Diniz-Filho et al. 2008). For all species, model diagnostics indicated a strong evidence of a single best model ($w_i > 0.9$; Anderson et al. 2001), which in all cases was the model containing all three variables. Since the model diagnostics indicated negligible extra explanatory power of the alternative models ($\Delta_i < 1$) compared to the full model, we used the full model for further analysis rather than using model averaging. We calculated partial and full MacFadden R^2 to evaluate model performance. A summary of the IT analysis is given in Appendix 3.

Case study area

The study area is located in the peat meadow district of the Dutch provinces of Utrecht and Zuid-Holland and covers about 440 km² (Fig. 1). Because the area is

situated in the delta of the river Rhine, peat deposits of several metres thickness are intermingled with many small sandy crevasse deposits. The major part of the area is in use for dairy farming. Because of the open character of the landscape and the relatively high groundwater levels, peat meadows have become an important habitat to meadow birds (Melman et al. 2008; Van der Vliet et al. 2014). About 15 % of the farms in the case study area receive grants to improve conditions for meadow birds (Vogelzang et al. 2009). Other land use types include urban areas and nature reserves (generally meadows or wetlands). The area is split up into hundreds of small sub-catchments with independent water level management. Most water levels are maintained at 30–70 cm below surface level, resulting in average soil subsidence rates of 5–10 mm year⁻¹.

Landscape change scenarios

Two landscape change scenarios were compared with the current situation. Both scenarios assumed that surface water levels are kept artificially low to

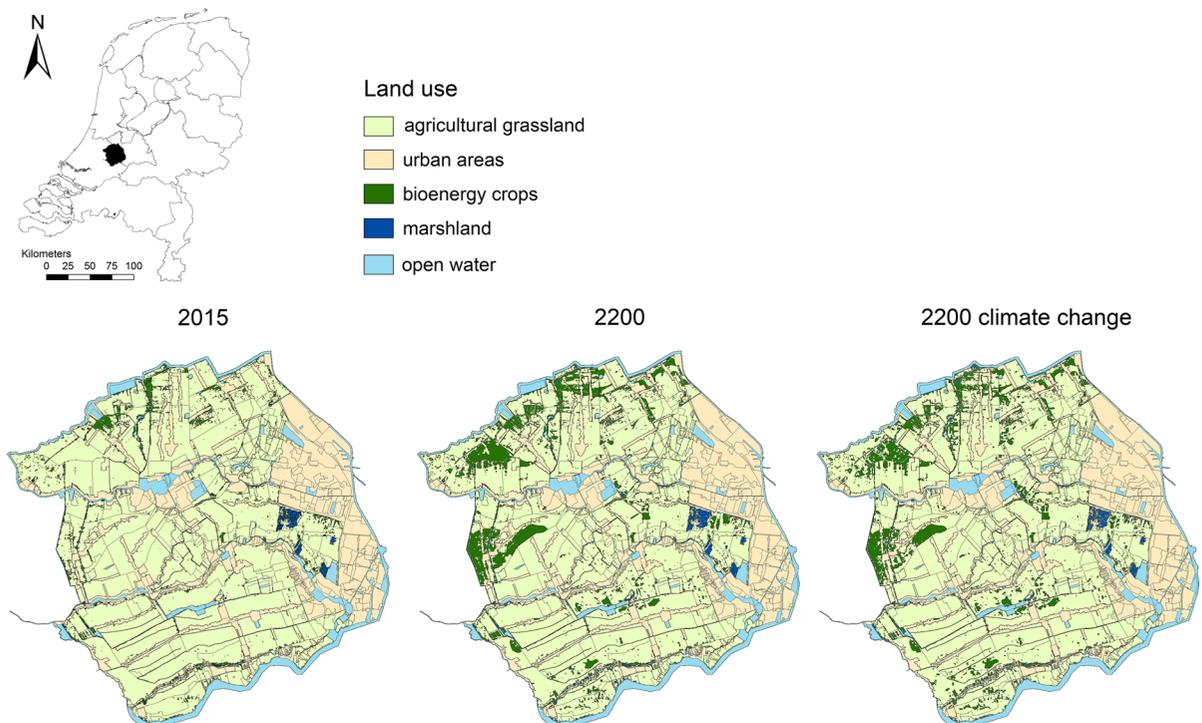


Fig. 1 Land use maps for the current situation (2015) and the two landscape change scenarios (2020) with and without climate change. The location of the study area in the Netherlands is indicated in the *top left* corner

facilitate agriculture, which leads to continued soil subsidence due to peat oxidation (Querner et al. 2012). This is the current water management strategy in the area. The first scenario considered changes in the landscape in response to soil subsidence and water management only, while the second scenario also included climate change effects. For climate change, we considered additional effects of changes in precipitation and evapotranspiration on groundwater levels, based on meteorological changes that are expected for the upper range of regional climate change projections (the W+ scenario cf. Van den Hurk et al. 2006). These result in lower groundwater levels and hence faster soil subsidence as time progresses. The additional effect of increased temperatures on the rate of soil subsidence was not taken into account. Because soil subsidence is considered as a major driver of potential land use change for peat meadow areas (Querner et al. 2012) and rates of soil subsidence are relatively low, we considered a much longer timeframe for our scenarios than is generally used in climate projections, in this case up to 2200. Although this timeframe may be of limited value to short term policy decisions, the empirical models used to generate our landscape change scenarios were developed by the water control board, responsible for water management in the study area. The models are used to guide sustainability of long term water management policies because of the high societal costs of soil subsidence due to difficulties in water management and damage to buildings and infrastructure. On the long term, local differences in soil subsidence rates and increased difficulties with adjusting water management to extreme levels of soil subsidence eventually lead to wetter soils, which can be aggravated by climate change.

Future groundwater levels were calculated with a model based on current groundwater levels, current soil profiles and an empirical relation to calculate the rate of soil subsidence that was developed for part of the study area (Hoogland et al. 2012) with 5 year time steps. Every 5 years soil profiles and surface water levels were adjusted to account for average soil subsidence, after which groundwater levels in all small contiguous sub-catchments were recalculated. Groundwater maps with a 25×25 m resolution were averaged to 500×500 m to match the grid cell size of our bird density models.

Agricultural land use in peat meadow areas is strongly dependent on groundwater levels, because high groundwater levels lead to yield reductions and hence income loss to farmers. We calculated expected yield losses based on our calculations of future groundwater levels, using empirical relations between groundwater level and yield (De Vos et al. 2006). Below a certain threshold of yield loss, dairy farming is no longer profitable and farmers will switch to other crops or abandon farming altogether, which leads to land use change. Based on economic data of farms in the study area, we estimated this threshold to be at 40 % income loss. At high groundwater levels, an alternative option is to switch from dairy farming to biomass crops for energy production, specifically willow coppice or reed (Londo et al. 2001; Kuhlman et al. 2013). These options are currently not attractive economically, but can be a viable option especially under global change scenarios with higher groundwater levels due to soil subsidence (which leads to income loss), increased fuel prices and increased societal demand for renewable energy sources (Kuhlman et al. 2013). We therefore assumed in our scenarios that at 40 % income loss due to higher water levels, land use would change from grassland to bioenergy crops. However, if spring groundwater levels became higher than 15 cm above the soil surface, we assumed that bioenergy farming is no longer possible and land use would change to marshland. Land use maps were produced at 25×25 m resolution. These were used to determine the dominant land use type for a 500×500 m grid to match the resolution of our bird density models.

A switch from dairy farming to bioenergy crops does not only lead to conversion of grassland into a land use type that is unsuitable for meadow birds, but the concomitant loss of landscape openness (Gaucherel et al. 2010) may also reduce the habitat suitability of the remaining grassland. Since meadow birds rely on actively deterring predators by displaying aggressive behavior, they require good predator visibility (Dyrce et al. 1981; Van der Vliet et al. 2008). Densities of breeding meadow birds have indeed been shown to decrease with decreasing openness of the landscape as a consequence of high vegetation (Van der Vliet et al. 2008, 2010). For both landscape change scenarios and the current situation we calculated landscape openness based

current topographical features, i.e., buildings and vegetation >2 m (based on the most recent vector map of topography from November 2013) and modeled areas with bioenergy crops (which we consider to be vegetation higher than 2 m), following Dijkstra and Van Lith-Kranendonk (2000): For each 500×500 m grid cell, we calculated total area percentages of buildings and vegetation higher than 2 m and converted these to a nine point openness scale (cf. Dijkstra and Van Lith-Kranendonk 2000; Appendix 2). We assumed these topographical features to remain intact unless land use changed. To avoid double counting, we removed any topographical features that overlapped with the new land use types of bioenergy crop or marshland, after which the openness for each grid cell was recalculated.

All GIS work was done in ArcMap 10.1.

Future relative bird densities

We calculated future relative bird densities with the regression models for each of the bird species, using the scenario maps of ASG level, land use and openness as input. We initialized the model with the most recent input maps for the study area and used the model outcome for 2015 as the baseline map for comparison with the landscape change scenarios (referred to as 'current situation' in the remainder of this paper). In this way, any observed differences between the baseline situation and the landscape change scenarios arise from changes in the explanatory variables in the model rather than from differences between initial data and model representation of that data.

To take the uncertainties in our models into account, we present maps based on the quartile ranges of relative bird densities rather than relative density itself, where we consider the upper quartile as core breeding areas for the different bird species (cf. Van der Vliet et al. 2014). We consider the conditions in these areas to comply best with the species habitat demands in the breeding season. Similarly, we consider the lower quartile (including null values) to represent the areas with the worst combined habitat conditions. We calculated the relative change in area covered by each quartile as an indicator of habitat suitability change for each scenario.

Results

Habitat suitability models

Our IT analysis (Appendix 3) identified the model including all three landscape variables (groundwater level, land use and openness) as the single best model for all species, indicating that all three landscape variables were important to explain relative bird density data. The relative contribution of specific landscape characteristics to relative bird density, however, differed between species (Table 1). Although in many cases the differences in R^2 values between landscape variables were small, groundwater levels consistently contributed least to relative bird density for all species. Land use contributed most to the explained variance of Black-tailed godwit and Northern lapwing density, while for Common redshank and Eurasian oystercatcher, openness contributed most to the explained variance by the model. For each species, the regression model based on the three landscape characteristics explained around 30 % of the observed variance in relative bird densities (Table 1).

Landscape change scenarios

Overall, ASG levels increased in both landscape change scenarios, although groundwater levels had increased slightly less in the climate change scenario. Because water levels are managed in several smaller contiguous areas, increases in groundwater levels differed locally, with generally larger increases in the northern and central-western part of the area. Our scenarios for 2200 indicated switches from dairy farming to bioenergy crops over larger contiguous areas for these locations, with the climate change scenario showing a more fragmented pattern (Fig. 1). In the rest of the study area, switching to bioenergy crops happened only locally. In both scenarios, switches to bioenergy crops were predicted for around 10 % of the current grassland area. Changes to marshland were few and scattered. The total area for each land use type is given in Table 2 for every scenario.

In the openness maps (Fig. 2) the larger contiguous areas where land use switched to bioenergy crops

Table 1 Partial and model MacFadden R^2 values for single best models of meadow bird densities

	Groundwater	Land use	Openness	Model
Black-tailed godwit	0.05	0.12	0.08	0.32
Common redshank	0.05	0.06	0.13	0.30
Eurasian oystercatcher	0.03	0.05	0.10	0.30
Northern lapwing	0.04	0.08	0.07	0.32

All relations between explanatory habitat characteristics and relative bird density were positive

Table 2 Total areas of each land use type in each scenario

	2015	2200	2200 climate change
Bioenergy crops	9.16	29.51	28.72
Grassland	285.99	265.27	266.04
Marshland	2.51	2.64	2.79
Open water	33.99	34.04	33.95
Urban area	108.58	108.77	108.74

Area is given in km^2 . Note that bioenergy crops are present in the baseline situation (2015), because we present modeled land use rather than actual land use for reasons of comparability with the landscape change scenarios

appear as the least open areas, while the more local changes led to reduced openness of the landscape in the remainder of the area. Urbanized areas appear as relatively closed landscape in all scenarios.

Future relative bird densities

Figure 3 shows relative bird densities for the current situation and the two scenarios. Black-tailed godwit, Common redshank and Eurasian oystercatcher show rather similar density patterns. In the current situation,

a large part of the area provides suitable breeding habitat for these species, with the exception of highly urbanized locations (compare Fig. 1). Core areas are located in the northern and south-western part of the area. In both landscape change scenarios, however, the quality of the breeding habitat relative to the current situation is reduced. Especially in the northern part of the area, much of the core habitat is lost, while the area in the south-west is less affected.

Table 3 shows that in total, these species lose just over 30 % of their core habitat (upper quartile) of the current situation in both landscape change scenarios, with a slightly higher loss in the scenario that includes climate change. Additionally, another 20 % of the area of the current second quartile of relative bird densities was lost. The total area with the least favourable habitat conditions (i.e. the lower quartile in the current situation) increased with 15–35 %, depending on the species.

Northern lapwing distribution in the area deviates from the three other species, with high densities over large contiguous areas in the northern part of the area. A large area with high densities is also found in the central-east, in between two highly urbanized locations (see Fig. 1). The south-western part of the area,

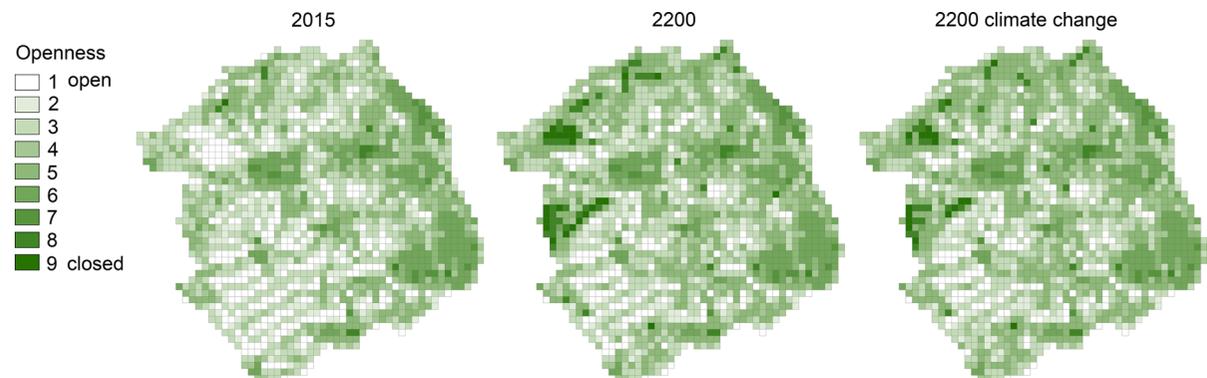


Fig. 2 Openness maps for the current situation (2015) and the two landscape change scenarios (2200) with and without climate change. Openness ranges from 1 (completely open) to 9 (completely closed)

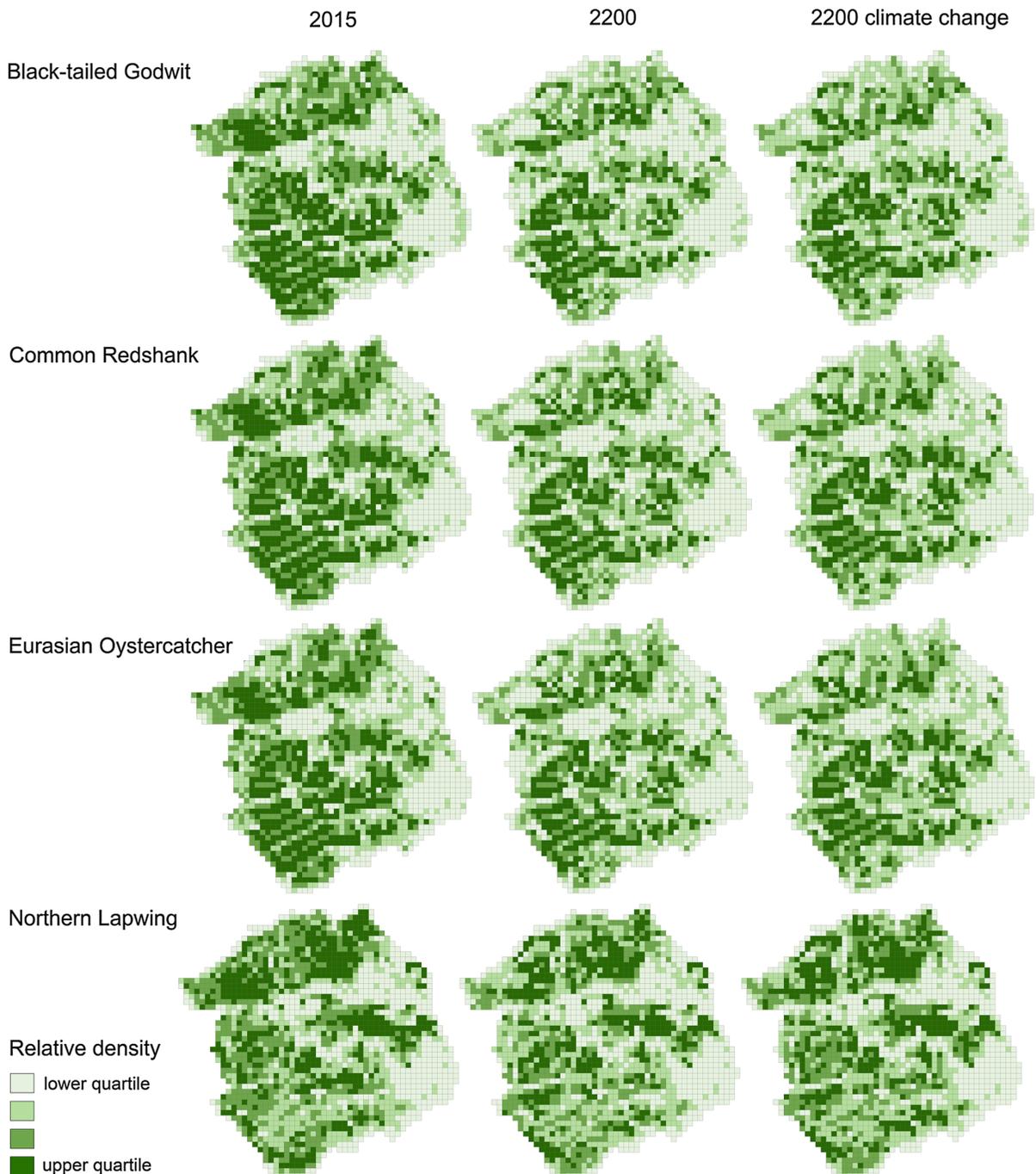


Fig. 3 Quartile ranges of relative densities of Black-tailed godwit, Common redshank, Eurasian oystercatcher and Northern lapwing for the current situation (2015) and the two landscape change scenarios (2020) with and without climate change

which shows high densities for the three other species, is less favored by Northern lapwing. This species shows the smallest response to both landscape change scenarios. The area with the best habitat (i.e., areas in

the upper quartile of relative densities) is reduced with 16.7 % in the scenario without climate change. In some parts of the area, specifically in the southwestern part, new habitat patches with an optimal

Table 3 Changes in habitat quality for the scenarios for 2200 without and 2200 with climate change (2000 cc)

	Godwit		Lapwing		Oystercatcher		Redshank	
	2200	2200 cc	2200	2200 cc	2200	2200 cc	2200	2200 cc
Q1	−32.8	−34.2	−16.7	−6.6	−31.9	−32.6	−31.9	−32.4
Q2	−22.4	−21.7	24.9	27.1	−18.8	−21.7	−19.2	−23.3
Q3	18.8	31.0	−17.0	−25.6	24.4	39.4	27.6	40.0
Q4	34.4	25.1	6.8	5.2	24.2	15.2	21.5	15.8

Changes are expressed as percentage area loss (negative numbers) or gain relative to quartile ranges of modeled relative bird densities in 2015 (with Q1 representing the highest and Q4 the lowest quartile)

combination of landscape characteristics emerge under both landscape change scenarios. Northern lapwing seems to profit from the additional climate change effects. The reduction in core habitat areas is only 6.6 % when climate change impacts are taken into account.

Discussion

In this paper we developed a set of regression models to assess the impact of changes in peat meadow landscapes in response to climate change on meadow bird habitat quality. Our main finding was that habitat quality was reduced for all meadow bird species in landscape change scenarios with and without climate change. With the exception of Northern lapwing, all species lost more than 30 % of their core habitat (habitat with the upper quartile of bird densities in 2015) in both scenarios. The main reason for habitat deterioration was the conversion of grassland to bioenergy crops that reduced landscape openness. No important direct effects of climate change on habitat quality were identified, except for Northern lapwing for which climate change slightly mitigated the impact of land use change. Our case study illustrates that direct impacts of climate change on the landscape play a minor role compared to human adaptation to environmental change. In contrast to our expectations, land use change was primarily brought about by soil subsidence in response to the sustained management of lower water levels and hardly driven by additional effects of climate change. These findings are in concordance with Riordan and Rundel (2014), who found that projected land use change was a more important driver of habitat loss for plant species than climate change in a Californian ecosystem. Because

our study focuses on the impact of climate change on local habitat quality and bird densities rather than range shifts, we did not include climate envelope modeling in our study, but restricted ourselves to habitat modeling only. We therefore ignore physiological and phenological adaptations of species to climate change, which may have additional effects on local population size (Mustin et al. 2007; Kleijn et al. 2010). Nevertheless, the importance of human adaptation to long term environmental change for future habitat quality and bird densities that we identify here supports the conclusions of Barbet-Massin et al. (2012) and Jongsomjit et al. (2013) that projections of the impact of climate change on bird species populations and distribution should include landscape changes and the impact of those changes on habitat quality in order to give a reliable indication of the vulnerability of species to climate change. Ignoring habitat loss due to land use change will in most cases lead to an underestimation of population changes under different climate scenarios, especially if landscape change leads to increased habitat fragmentation (Travis 2003).

A main factor driving land use change in our study area were increased groundwater levels due to soil subsidence. We found little influence of climate change on groundwater levels in the study area, where water levels are highly managed by humans. Our model outcomes suggest that in such highly managed landscapes, the impact of climate change on groundwater levels can be accommodated by adaptations in water management. In our scenarios, however, we did not consider the possible accelerating effects of increase temperatures on soil subsidence, making our estimate of soil subsidence rates rather conservative. Inclusion of those temperature effects may lead to higher soil subsidence rates (Querner et al. 2012),

which may result in even larger areas where dairy farming will no longer be profitable because of increased groundwater levels than we currently modeled. Although high groundwater levels are positive for meadow birds because they increase food accessibility, they are detrimental to grass yields. These yield losses may become so large, that continuation of dairy farming is no longer profitable. Our income loss calculations showed that this is the case for larger contiguous areas of current grassland in the northern and western part of the study area, and a large number of smaller areas in other locations.

If dairy farming is no longer profitable due to high groundwater levels, this may lead to land use change. In peat meadow landscapes, growing biomass crops for energy production, such as willow coppice or reed, can be an alternative form of income for farmers (Londo et al. 2001; Kuhlman et al. 2013). This practice is currently not profitable, but the profitability of these alternatives is highly dependent on prices of agricultural produce compared to fuel prices, CO₂ emission prices and the societal demand for more sustainable energy sources (Dale et al. 2011; Kuhlman et al. 2013). It may therefore become an attractive land use option under scenarios of global change that include considerable climate effects (Gaucherel et al. 2010). In this paper, we therefore assumed a switch to bioenergy crops in those locations where dairy farming is no longer profitable. We chose a rather arbitrary 40 % income loss as a threshold for switching to alternative sources of income, based on expert judgment of the current economic situation of several farms in the study area. Because dairy farming is currently more profitable than growing bioenergy crops, there are no empirical data available from farmers who have switched to better inform this choice. With the current assumptions, our empirical models predict a switch to bioenergy crops for about 10 % of the current grassland area, which equals 7 % of the total study area. The increase is slightly lower when climate change is taken into account, because of the lower groundwater levels (and hence lower yield losses) due to climate change (Table 2; Fig. 1).

Conclusions

We conclude that indirect effects of climate change (i.e., land use and landscape changes) should be

considered in long term conservation planning, especially under scenarios of environmental change that may lead to changes in agricultural practices. We have shown here that the effects of human adaptation to environmental change rather than direct impacts of climate change on habitat quality were driving loss of habitat quality in a highly managed landscape. Such indirect effects of climate change are currently largely ignored in conservation schemes, but may have significant effects for population dynamics (see also Kleijn et al. 2010). Meadow bird conservation schemes, for instance, generally focus on adjustments of local field management only (Verhulst et al. 2007). This study suggests that such local measures only contribute to long term conservation aims if they take the wider landscape ecological setting into account and if they are embedded in a large scale spatial planning policy that regulates land use change. Simple habitat quality models like the ones we present in this paper may aid the development of such a planning policy.

Acknowledgments JvD was financially supported by the Climate Adaptation for Rural Areas (CARE) Project, which was funded by the Knowledge for Climate Programme (<http://knowledgeforclimate.climate-research-netherlands.nl/climate-adaptation-for-rural-areas>). SOVON Vogelonderzoek Nederland provided the Dutch relative bird density datasets. Harry Dijkstra, Jetty van Lith-Kranendonk and Jaco van der Gaast (all Alterra Wageningen) provided the openness and groundwater datasets. Mara Baudena kindly assisted with programming the reclassification of the openness maps. Paul Opdam gave very useful suggestions for the presentation of the manuscript.

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