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Original Research Article

Reed cut, habitat diversity and productivity in wetlands

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ABSTRACT

There is a conflict between nature conservation and thatching industry regarding the management of reedbeds. On one hand, reedbeds are of an economical importance by providing thatching material, on the other hand, they harbour several endangered species. Reedbeds are typically managed by winter cutting, but its impacts on biodiversity are poorly understood. Our aim was to study the effects of winter cutting on the habitat diversity and structural heterogeneity of wetlands in a lowland alkali landscape (East-Hungary). We tested the following hypotheses: (i) Both diversity of plant species and habitat diversity are lower in winter cut wetlands compared to unmanaged stands. (ii) The distribution of biomass (green biomass, litter and standing dead biomass) is more homogeneous in winter cut wetlands compared to unmanaged ones. We found that winter cutting decreased habitat diversity and structural heterogeneity at multiple scales. Number of plant species and all measures of habitat diversity (number of patches, vegetation types and the length of vegetation margins) had lower scores in cut wetlands than in unmanaged ones. We found that unmanaged wetlands harboured high amount of accumulated biomass and they also maintained high habitat diversity likely due to the heterogeneous distribution of the biomass. In unmanaged wetlands, biomass accumulation did not decrease habitat diversity and also contributed to a higher structural heterogeneity. In cut wetlands, expansion of reed was an important driver of the decrease in habitat diversity and structural heterogeneity. Reed expansion likely overrode fine-scale edaphic conditions (hydrology and salinity) in shaping vegetation patterns; thus we suggest to avoid intensive winter cutting.

1. Introduction

Even wetlands harbour a considerable species- and habitat diversity, they are among the most endangered habitats in the world (Mitsch, 2005). Common reed (*Phragmites australis*) often forms extended stands in European wetlands (Valkama et al., 2008). On one hand, reedbeds provide thatching material which makes them important for economy. On the other hand they also represent important habitats for plants, birds and invertebrates, including many rare and endangered species (Valkama et al., 2008; Zlinszky et al., 2012). Even though it would be crucial to preserve the remaining natural wetlands and ensure their sustainable conservation management, there is often a conflict between nature conservation and thatching industry regarding the proper management of wetlands (Trnka et al., 2013). The most frequently applied management measures comprise cutting, burning and grazing (Valkama et al., 2008); however some studies pointed out that wetland diversity might be maintained merely without any management (Wanner et al., 2014).

In wetlands characterised by reed cutting is the most typical management type. Companies generally cut the reed in wintertime (annually or in every second year) for the thatching industry. Winter cutting provides a good quality reed with thick and dense shoots (Ostendorp, 1999). This kind of management effectively rejuvenates reed stands by removing dead biomass (Poulin and Lefebvre, 2002) and enhances the vegetative expansion of the reed resulting in homogenous vegetation structure (Engloner, 2009). Winter cutting can have detrimental effects on species and habitat diversity by decreasing the inner structural heterogeneity of the reed stands and eliminating the mosaic of young and old stands (Ostendorp, 1999; Poulin and Lefebvre, 2002; Trnka et al., 2013). Shifts in species composition, such as the encroachment of competitor grasses can cause decreased species- and habitat-diversity (Hejcman et al., 2009; Marrs et al., 2004; Mauchamp

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et al., 2001). Reed, which is a typical competitor grass species of the wetlands can effectively suppress other wetland plant species by competitive exclusion via reduced light availability (Ungar, 1998) and increased resource competition (Wang et al., 2012). High competitiveness and rapid spreading of reed made it an invasive plant species in North-America (Howard et al., 2008) which causes serious nature conservation problems also in other parts of the World (Silliman and Bertness, 2004).

Structural changes induced by winter cutting might have a negative effect on the populations of passerine birds due to the decrease in their prey abundance and the availability of proper places for nesting and the increase in the risk of nest predation (Graveland, 1999; Poulin and Lefebvre, 2002). Several studies found that abundance and species number of arthropods decreases due to winter cutting as it destroys overwintering individuals, decreases the structural diversity of the habitats and increases the fluctuations in microclimate (Schmidt et al., 2005; Valkama et al., 2008).

In wetlands, habitat diversity and structural heterogeneity are crucial drivers of ecosystem functioning and sustaining biodiversity (Nolte et al., 2014; Rahbek and Graves, 2001). Habitats with high compositional and structural heterogeneity can provide various niches for species co-existence (Pacala and Tilman, 1994). Habitat diversity can be expressed by the number, size and Shannon diversity of vegetation patches, while the spatial distribution of biomass fractions is a good measure of structural heterogeneity (Kallimanis et al., 2008; Kohn and Walsh, 1994; McCoy and Bell, 1991). The aim of our study was to test the effects of intensive reed management represented by annual winter cutting on the habitat diversity and structural heterogeneity of wetlands. We tested the following hypotheses: (i) *Habitat diversity hypothesis*. Both diversity of plant species and habitat diversity are lower in winter cut wetlands compared to unmanaged stands. (ii) *Structural heterogeneity hypothesis*. The distribution of biomass (green biomass, litter and standing dead biomass) is more homogeneous in winter cut wetlands compared to unmanaged ones.

2. Materials and methods

2.1. Study sites

Our study sites are situated in the Hortobágy National Park (East-Hungary; N 47°30′ E 21°12′) and involved in the Natura 2000 network as Hortobágy Special Area of Interest (HUHN20002). The National Park was designated to preserve one of the largest connected open landscapes in Europe, characterized by a mosaic structure of alkali steppes, meadows and wetlands (Deák et al., 2014a,b). Wetlands comprise both inland salt- and freshwater marshes.

We studied four wetlands: two of them ('Csattag' and 'Bőgő-lapos') have been annually cut by machinery in wintertime (December–February) since the 1960s, and two of them ('Csépa' and 'Kecskeri') were left unmanaged. The studied wetlands had no connection with each other; the minimum distance between them was 4 km. Each studied wetland was characterised by a mosaic of alkali and non-alkali wetland vegetation. The average depth of the water cover ranges between 5-30 cm in springtime and generally the wetlands dry out completely till the end of summer.

2.2. Vegetation types

Natural wetland systems in the study region are generally characterised by a high compositional diversity and structural heterogeneity. The pattern of the different vegetation patches is generally driven by plant–soil feedbacks resulting in a self-organized patchiness. Several vegetation types can co-occur in a diverse mosaic structure; their spatial pattern is driven by edaphic conditions such as soil salinity and hydrology (Deák et al., 2014a). The studied wetlands harboured alkali and non-alkali marsh vegetation, sedge vegetation, smaller patches of open muddy surfaces and aquatic vegetation (see Appendix 1). Alkali marsh vegetation included alkali reedbeds (*Phragmites* marsh), *Bolboschoenus* marshes and *Schoenoplectus* marshes. Non-alkali marshes comprised *Glyceria maxima* marshes, *Typha angustifolia* and *T. latifolia* marshes. All of these vegetation types were characterised by a high vegetation cover and biomass. Sedge vegetation was characterised by *Carex vesicaria* and *C. riparia*. Muddy surfaces were characterised by a low vegetation cover and biomass; where the characteristic species were *Agrostis stolonifera*, *Echinochloa crus-galli*, *Lycopus exaltatus* and *Rumex palustris*. Aquatic vegetation was present only in very small patches characterised by *Utricularia vulgaris*. We found patches in which the most abundant species of different vegetation types co-occurred and composed mixed stands of the above mentioned vegetation types. We classified *Phragmites* and *Typha angustifolia* stands into 'young' and 'old' groups based on the amount of dead biomass (including litter layer and standing dead biomass).

2.3. Sampling design and data collection

We designated a 100 m $\times$ 100 m study site in each wetland. Prior to the field surveys we performed own flight campaigns for acquiring high-resolution and up-to-date georeferenced aerial photos of the study sites. The photos were taken two weeks before the field survey. During the field survey in July 2011 we compiled the vegetation map of the study sites. We considered a unit as a separate vegetation patch if it could be recognised in the aerial photos at a scale of 1:200, had discrete borders and its vegetation and structure was different from the neighbouring units. Vegetation maps were digitalized in QGIS 2.0 (QGIS Development Team, 2014).

From each vegetation type, three 50 cm $\times$ 50 cm sized above-ground biomass samples were collected in each study site. Biomass samples were dried, then sorted to green biomass, litter and standing dead biomass. Green biomass was further sorted to species. We weighted the sorted samples with an accuracy of 0.01 g.

2.4. Data analysis

For further analyses we randomised ten 10 m $\times$ 10 m plots within each 100 m $\times$ 100 m study sites using QGIS. We calculated the number and area of each vegetation patch and the length of vegetation margins inside each randomised plot. We calculated the Shannon diversity of the vegetation types in the 10 m $\times$ 10 m plots. We also calculated the average total biomass scores for every vegetation type based on the biomass samples. We calculated the amount of green biomass, litter, standing dead biomass, total dead biomass and total biomass scores for each plot. We also calculated the coefficient of variation (CV; standard deviation divided by the mean) for the biomass scores inside the plots.

For exploring the effects of management (fixed factor) on dependent variables we used General Linear Models (GLM) using SPSS 20.0. We used the number of patches, number of vegetation types, diversity of patches, diversity of vegetation types, length of the vegetation margins, coefficient of variation (CV) of green biomass, litter, standing dead biomass and the total amount of biomass as dependent variables. Location of the study sites were considered as weighting factor. To analyze the relationship between habitat heterogeneity (expressed by the Shannon diversity of vegetation types) and structural heterogeneity