RESTORATION AND MANAGEMENT OF RIVER FLOODPLAINS:

EXPERIENCE OF THE LIFE+ PROJECT DVIETE

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Restoration of Corncrake habitats in Dviete floodplain Natura 2000 site
2010-2015

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Restoration and management of river floodplains: experience of the LIFE+ project DVIETE

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Content

Results of the hydrological monitoring of the Dviete floodplain

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Summary

This paper summarises the groundwater level observations that were performed within the Dviete floodplain area under the LIFE+ project "Restoration of Corncrake habitats in Dviete floodplain Natura 2000 site", as well as hydrological observations of the surface and subsurface water levels that were performed by scientists and students of Daugavpils University and the Latvian University of Agriculture under their research activities, and by local researchers voluntarily. The analysis of the obtained results indicates a close correlation between the hydrological regimes of the surface and subsurface waters within the Dviete floodplain area. During the spring floods, the height of the water level in the Dviete floodplain is significantly influenced by its hydrological connectivity to the Daugava River. In contrast, during the summer and winter low water periods, the local weather conditions that regulate formation of the surface and subsurface runoff are more important. Results and conclusions described in this paper must be taken into account when the effectiveness of restoration measures of the Dviete River meanders conducted under the LIFE+ project are assessed and future actions planned in relation to the protection of the Dviete floodplain meadows and management of their soil moisture level in summer.

Keywords: Dviete floodplain, groundwaters, water level variation, weather, Daugava

Introduction

The history of hydrological observations in the Dviete floodplain area can be divided into three main steps.

1. The Dviete floodplain area was first noted by hydrologists during the disastrous floods in 1931. Already in 1932 there were two new hydrological posts established at Līksna and Nīcgale villages upstream and downstream from the Dviete floodplain, and daily water level observations were started there (Stakle, Kanaviņš 1941; Анон 1987). Meanwhile, a large-scale draining campaign was started within the Dviete floodplain area itself, which included straightening of the Dviete River and draining of the floodplain lakes located there. The consequences of these activities can still be observed in nature (Račinskis 2005).

Nevertheless, special hydrological posts were not established within the Dviete floodplain area until the 1960-ties. The first hydrological post was only established on the Dviete River at Dviete village in 1967. However, it functioned continuously for 10 years (1968–1978), and its data records were partly published in 1987 (Анон. 1987).

In order to assess the size of agricultural lands that are inundated at certain floodwater levels in spring, hydrological observations at Dviete village were also performed in 1981-1986 (Anon. 1987). Short-term hydrological observations were also performed at the upland reach of the Dviete River in 1952–1955 (Zavickis, Gruberts 1986), as well as near Skuķu Lake in 1999 (Gruberts et al. 2005).

2. The second step in the history of hydrological observations of the Dviete floodplain started in March 2005 when researchers and students from Daugavpils University (DU) started regular water level measurements from the bridges across the Dviete River at Bebrene (the so called 'Sloboda Bridge') and Dviete. These observations were performed as part of the DU research project *„Seasonal dynamics of hydrobiological parameters of Daugava's floodplain lakes"* (Gruberts 2006). Meanwhile, the first DU automatic weather station was established on the right side of the Dviete valley in Putnusala village at "Atāli" farmstead. Since then, this station records the main weather characteristics (air temperature, precipitation, humidity etc.) of the Dviete floodplain area on an hourly interval (Gruberts 2013a).

Hydrological observations from the bridges across the Dviete River were continued until April 2009, and their results were described and discussed in bachelor and master theses of the environmental science students at DU (Uļjans 2009; Uļjans 2011), as well as published in several scientific papers (Gruberts 2007; Paidere et al. 2007; Paidere 2008; Škute et al. 2008). Regular water level measurements from the Sloboda Bridge were restarted in 2011 and continued for another three years (Gruberts 2014b) thanks to the volunteer activity of Ārija Gruberte, a local history and nature explorer and former geography teacher at Bebrene parish.

3. The third step in the hydrological monitoring of the Dviete floodplain started in 2007. It was related to the first hydrological assessment of the possible renaturalisation of the riverbed of Dviete ordered by the Latvian Ornithological Society. This assessment was part of the project *„Management and renaturalisation of the Dviete floodplain nature park"* (2007-2008) financed by the Royal Bird Protection Society, the Netherlands (*Vogelbescherming Netherlands*). During this project, four permanent groundwater monitoring wells and surface water level monitoring lath were established on the right side of the Dviete River near Putnusala village (Fig. 2), and weekly water level measurements were started there (Indriksons 2008; Indriksons 2010).

The network of groundwater monitoring wells within the Dviete floodplain area was significantly enlarged in 2012 as part of the LIFE+ project „ Restoration of Corncrake habitats in Dviete floodplain Natura 2000 site". 15 new monitoring wells were established close to the old ones across the floodplain area (Fig. 2). Since then, the groundwater levels have been measured there four times per month. Measurements were also continued at the old monitoring wells established earlier. Their records, combined with other hydrological data series from the nearby hydrological posts, were used to assess the efficiency of the renaturalisation measures applied to the hydrographic network of the Dviete River during the LIFE+ project mentioned above. The main aim of this report is to provide overall insight into the hydrological observations performed within the Dviete floodplain since 2005. Its aim is also to state the importance of the Daugava's hydrology and local weather conditions in the seasonal and annular variation of the surface and subsurface water levels in the Dviete floodplain before the renaturalisation of its hydrographic network upstream from the Skuķu lake in January–February of 2015.

Study area

The majority of hydrological observations summarised in this article were performed within the Dviete floodplain or the **Dviete ancient valley**, a valley-like depression located between the Augšzeme Upland and the East Latvian Lowland (Fig. 1.) In fact, it is not very correct to call it the „Dviete floodplain" because

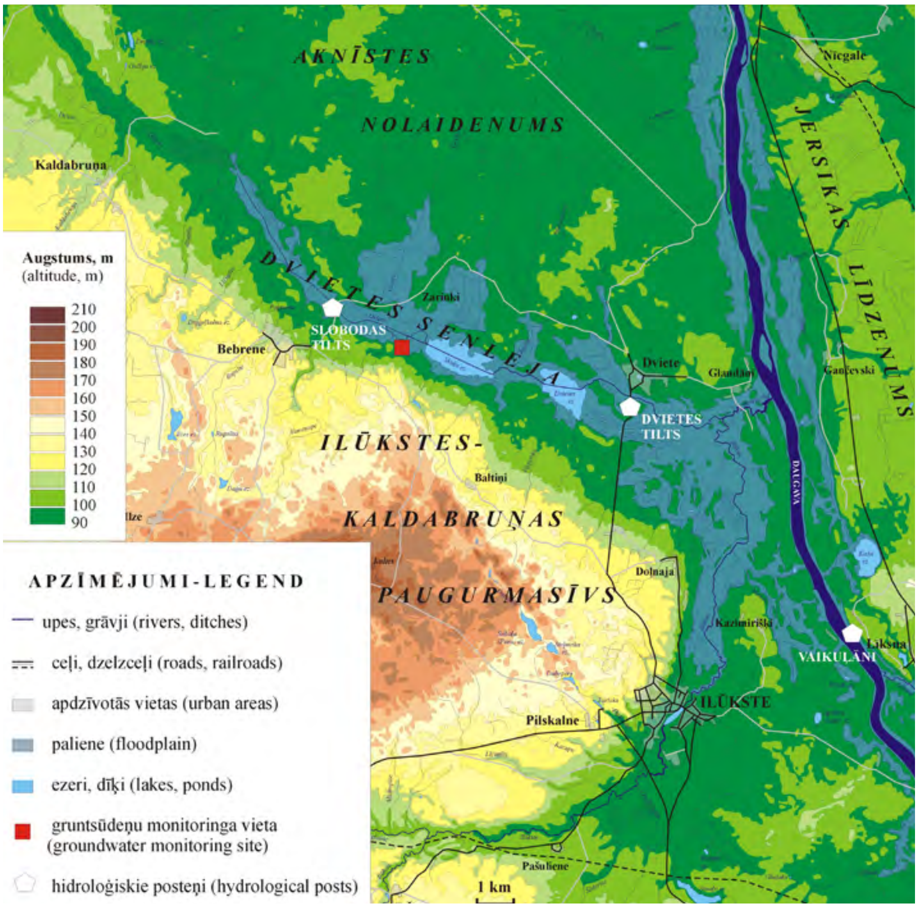

Figure 1. *Location of the Dviete ancient valley within a physiographic map of the South-East part of Latvia (inundated area at the long-term mean flood level (90 m a.s.l.) is presented in greyish blue colour)*

its hydrology, especially the boundaries of inundation and the flood water level height, is controlled by the Daugava River (Gruberts 2004).

Hypothetically, the Dviete ancient valley is part of the ancient buried hydrographic network of former river valleys that was carved into the surface of Devonian bedrocks (sandstones, silts, etc.) and filled by later (Quaternary) deposits – sand, clay, peat, etc. (Эберхард 1972). The present shape of the Dviete ancient valley was also formed by active geological processes at the end of the last Ice Age, especially the movement of glaciers and their melting waters in this area (Gruberts, Soms 2004).

The central part of the Dviete ancient valley is occupied by an accumulative floodplain with Skuķu and Dviete lakes at its lowest depressions (Fig. 1). These floodplain lakes are of glacial origin and are among the largest floodplain lakes in Latvia (Gruberts 2003; Gruberts 2006). The river Dviete (also called the 'Sloboda' in its upland reach and the 'Berezovka' near its mouth) flows through them and connects these lakes with the Daugava River at the Glaudānu Island, where large sandbars are located. Regular ice jams are formed here during the spring floods. Therefore the Daugava's floodwaters are forced to enter the Dviete valley (Глазачева 1965).

The hydrological regime of the Dviete ancient valley (or the Dviete floodplain in further text) is quite complex. The peak flood discharges of the Dviete River itself are determined by the hydrological and meteorological processes within its drainage area (i.e. depth of the snow cover, amount of precipitation, evaporation intensity, soil moisture, development of the surface and groundwater runoff, etc.), as well as by the water exchange with the Daugava River. For the spring floods with a 10 % reoccurrence probability, the maximum discharge of the Dviete River at the Dviete Bridge can reach 70 m^3 per second (Gruberts, Zavickis 1986). However, almost every year at the beginning of the floods it is counteracted by a reverse floodwater flow from the Daugava River side. Theoretically, its maximum discharge at the Dviete Bridge can reach 399 m^3 per second, therefore exceeding the maximum direct discharge from the local drainage area almost sixfold. Even at the Sloboda Bridge the maximum reverse discharge of this flow can reach 120 m^3 per second (Zavickis, Gruberts 1986).

Usually, an intrusion of the floodwaters into the Dviete floodplain begins at the end of March, simultaneously with the ice drift in the Daugava River, and continues until mid-April. Extremely high floods have been recorded here in those years when massive ice jams are formed in the Daugava riverbed at the Glaudānu Island. The highest recorded floodwater level was observed in April-May, 1931. In the Dviete floodplain, its absolute height reached 94.22 m above sea level (a.s.l.). This was stated during the 1980-ties by technical levelling of this historic floodwater level according to the testimony of Jānis Kolosovskis – an eyewitness to those historic floods who lived in the Putnusala village near the floodplain border all his life. Very high floodwater levels were also recorded here in 1951 and 1956 (Zavickis, Gruberts 1986).

This floodplain area also has a very important role in the regulation of the hydrological regime of the Daugava River between Daugavpils and Jēkabpils. The results of complex geospatial analysis performed at the Department of Geography, Daugavpils University, showed that at the average floodwater level a 55 km long water body between Daugavpils and Jersika forms, with an average depth about 1.5 m, a total surface area of about 200 km^2, and a maximum floodwater storage capacity of about 310 million m^3 (Škute et al. 2008). However, recent statistical analysis of hydrological data records obtained at the Daugavpils and Jēkabpils hydrological posts shows that during the record high floods the Middle Daugava floodplain is capable of intercepting more than 192.7 million m^3 of floodwaters per day, and it can accumulate at least 0.62 km^3 of them for five consecutive days as was stated for the spring floods of April 1924 (Gruberts, Vilcāne 2015).

During the summer low water period, the surface runoff from the Dviete drainage area decreases dramatically due to the continental climate and dense network of drainage ditches. A certain minimal water level in the straightened channel of the Dviete River is maintained by beaver dams and dense aquatic vegetation in the overgrown Skuķu and Dviete floodplain lakes (Indriksons 2008), as well as by the Daugava's water level stage at the Berezovka River mouth. The ground's surface around these lakes is located only 2.5–3.0 m above the Daugava's mean summer water level here. Therefore, any medium sized water level increase in the Daugava River in summer, autumn or winter results in an inundation of the lowest parts of the Dviete floodplain area, especially the depressions of floodplain lakes Skuķu and Dviete. These two lakes belong to the group of repeatedly inundated floodplain lakes of the Middle Daugava River since they are flooded not only during the spring floods but also during the highest floods observed during the summer–autumn low water periods due to intense rainfalls in the Daugava's drainage area (Gruberts 2006; Gruberts et al. 2007).

Materials and methods

Measurements of the surface water levels

Regular measurements of the surface water levels in the Dviete River were started in March 2005 in order to assess the role of the hydrological regime of the Middle Daugava River in the ecology of phytoplankton

Figure 2. *Location of the groundwater level monitoring wells in the Dviete floodplain (map data © 2015 Google)*

and zooplankton communities of its floodplain lakes (Gruberts 2006; Paidere 2012). Measurements were performed 1–4 times per month at the bridge near Bebrene village (the so called 'Sloboda Bridge') and, also, at the bridge near Dviete village. Relative heights of water levels were measured in relation to the bridge's surface by a glass-fibre measuring tape 10 m in length with a 0.5 kg weight attached to its free end. The obtained relative heights were then recalculated to the absolute heights according to the specifications of both bridges available at the project archive of the A/S "Ceļuprojekts" (Zavickis, Gruberts 1986). In this way, water level measurements were performed regularly until March 2009. The accuracy of the measurements was +/- 1 cm.

To characterise the dynamics of the floodwater masses of the Daugava River within the Dviete floodplain area, as well as to obtain additional information on fluctuation of the surface water levels at Bebrene, regular measurements of the surface water levels were restarted at the Sloboda Bridge in March 2011. The same methods were used as described above. These measurements were performed until the end of 2013 (Gruberts 2014b).

The groundwater level measurements

Regular observations of the groundwater levels in the Dviete floodplain were started in the winter of 2007/2008. On December 15, 2007, four groundwater monitoring wells were established close to the old riverbed of the Dviete at the Putnusala village (Fig. 2). Plastic tubes 110 mm in diameter were used to construct the monitoring wells. Three wells were placed at the corners of a regular-sided triangle, oriented to the North, East and West. The distance between them was 10 m. The fourth well was placed in the centre of the triangle. The wells at the corners were drilled to a depth of 1.6 m below the ground surface with the tops of the tubes standing 33–43 cm above it. The central well was drilled to a depth of 3.34 m with the top of the tube standing 66 cm high above the surface. The side walls of the tubes were perforated to maintain free movement of the groundwaters through them. In order to stop the filling of the wells by sand and silt, the tubes were covered by a special fabric from the outside (Indriksons 2008). The absolute heights of the monitoring wells were stated by their instrumental levelling in relation to the nearby altitude mark located at the „Luksti" farm in the Putnusala village. Its absolute height is 101.056 m a.s.l. (A. Indriksons, pers. comment).

Since the establishment of these monitoring wells, the groundwater levels were measured there once per week. Until the end of 2012, these measurements were performed by Aija Kriškijāne (Kudiņa) – an employee of the Bebrene parish municipality.

To assess the possible impact of the Dviete River restoration measures on the floodplain biotopes and groundwater levels, 15 new monitoring wells were established nearby in 2012. They were located about 100 m away from the above mentioned wells to the East-Southeast and lined in a straight transect that is perpendicular to the Dviete River channel. The wells were drilled at regular intervals 20 m apart from each other. The transect line crosses the right side of the floodplain area of the Dviete River, up to its natural border at the foothills of the Putnusala hill (Fig. 2, 3).

The new monitoring wells were also constructed from plastic tubes 2 m in length and 110 mm in diameter, which were additionally fixed by metallic sticks at a depth of 0.7 m in order to reduce their vertical movement under the pressure of the ice cover.

Figure 3. *The new groundwater monitoring wells in January 2012 (Photo: D. Gruberts)*

Besides, they were also fenced-in to preserve their top ends from the impact of the grazing animals that are located here (Fig. 3).

In January 2012, the absolute heights and geographical positions of all the new monitoring wells were stated. The absolute heights of the wells were stated by their instrumental levelling to the above mentioned altitude mark at the „Lukstu" farm in the Putnusala village (A. Indriksons, pers. comment). The geographical coordinates of the new wells were recorded on January 29, 2012, by using a GPS device „Garmin 76". Near the 4th monitoring well, the surface water measurement lath was also fixed with its top end 4.31 m above the ground surface (Fig. 3, A. Indriksons, pers. comment). Regular monitoring of the groundwater levels in the new wells was started on January 16, 2012. The groundwater levels were measured four times per month by using a metallic measuring-tape that was lowered down to the groundwater surface. The relative heights of the groundwater levels were recorded in relation to the top ends of the tubes at first, and their absolute heights were recalculated afterwards in the MS Excel program by taking into account the results of the instrumental levelling mentioned above.

In May and September 2013, the relative heights of the tubes' top ends were measured in relation to the ground surface again. It was necessary because the wells were affected by a massive ice cover that was formed in the Dviete floodplain area at the beginning of winter 2012/2013. In January 2013, the pressure of the ice cover pushed the tubes many centimetres into the ground. Therefore, their absolute and relative heights were modified, and it was necessary to measure them once more in order to compare the previously obtained groundwater level data records to the newer ones.

In order to highlight the main factors that determine the hydrological regime of the surface waters and groundwaters of the Dviete floodplain, the data records obtained *in situ* were compared to each other, as well as to the water level records from the hydrological post „Daugava-Vaikuļāni", which are available in digital format at the Latvian Environment, Geology and Meteorology Centre (www.meteo.lv). For statistical analysis and interpretation of the results of the hydrological monitoring, the daily weather data records were obtained from the Daugavpils University weather station „Putnusala" located nearby (Fig. 2). The daily weather data records (precipitation, temperature) were obtained from Part IV-X of the Annuals of this station that were published recently (Gruberts 2013a-e; Gruberts 2014a; Gruberts 2015), as well as from the unpublished digital data records, which are available at the Department of Geography, Daugavpils University.

By using these data records, the decadal sums of precipitation and mean decadal air temperatures were calculated for the time period starting from January 2008 till August 2015. For statistical analysis of the obtained hydrological and meteorological data records the simplest linear correlation and regression methods were used.

Results and discussion

Results of the surface water level monitoring

During the Dviete River water level observations performed in 2005–2009 and 2011–2013, there were 10–11 distinct flood pulses recorded (Table 1), which coincided with the most significant water level fluctuations of the Daugava River at Vaikuļāni (Fig. 4, 5). During this period, the highest water level in the Dviete floodplain was observed on April 23, 2013, at the Sloboda Bridge (91.67 m a.s.l.). Relatively high water levels were also observed in 2011 and 2012. In other years the highest annual water levels didn't rise above the long-term mean highest floodwater level for this area (i.e. 90 m a.s.l.).

Usually, the highest annual water levels in the Dviete floodplain are observed in mid-April or in the second part of April, less frequently – in March and May. In 2005, the spring floods in the Dviete floodplain were relatively low, with the highest water level about 89 m a.s.l. In mid-May, they were followed by a record-high wave of flash floods caused by intense rainstorms in the Daugava drainage basin. During the flash floods, the highest water level at the Sloboda Bridge was about 1 m above that observed during the spring floods (1. tab.; Gruberts 2006). A similar situation was observed in August and September 2006, when the flash floods caused by rainstorms resulted in a quick water level rise in the Daugava valley and the Dviete floodplain up to the height of the spring flood level. In December 2012, after prolonged rainstorms, the water levels at the Sloboda Bridge were also above those observed during the spring floods and flash floods of 2006 (Table 1).

Table 1. *Annual heights of the floodwater levels in the Dviete floodplain, 2005–2013 (after unpubl. data of the Department of Geography, Daugavpils University, and Ā. Gruberte)*

Monitoring site	Dviete Bridge		Sloboda Bridge	
Year	Data	Level, m a.s.l.	Data	Level, m a.s.l.
2005	IV.10	89,12	IV.24	88,68
	V.15	89,90	V.15	89,66
2006	IV.17	88,43	IV.17	88,18
	IX.13	88,14	VIII.08	88,05
2007	III.14	89,06	III.14	88,78
2008	IV.25	88,12	IV.11	87,97
2009	IV.18	89,20	IV.18	88,80
2011	---	---	IV.15	90,47
2012	---	---	V.02	89,61
	---	---	XII.02	88,37
2013	---	---	IV.23	91,67

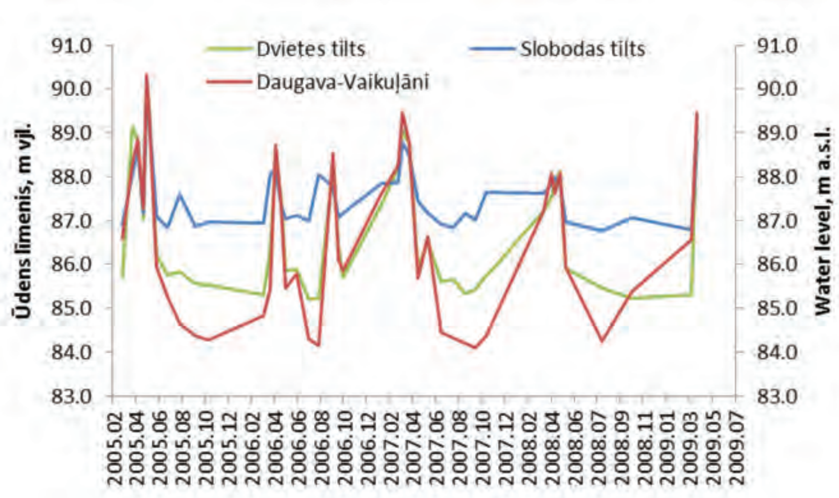

Figure 4. *Water level variation at the Dviete floodplain and the Daugava River at Vaikuļāni, 2005–2009 (after unpubl. data of the Department of Geography, Daugavpils University, and Latvian Environment, Geology and Meteorology Centre)*

The results of the regular water level observations in the Dviete River in 2005–2009 show that the highest water levels at both monitoring sites (i.e. at the Dviete and Sloboda bridges) almost coincide with the annual peaks of the floods in the Daugava River at Vaikuļāni (Fig. 4). However, the absolute heights of the peak floodwater levels are not equal. In most cases, the peak floodwater level in the Daugava at Vaikuļāni is about 26 cm higher than in the Dviete River at the Dviete Bridge, which in turn is about 26 cm higher than at the Sloboda Bridge located further upstream. By taking into account the distance of 9 km between both bridges, there is a marked slope (3 cm km^{-1}) in the water table in the region of the Skuķu and Dviete floodplain lakes that is oriented in the opposite direction to the 'normal' water flow in the Dviete River. Its existence is confirmed by regular observations of the reverse floodwater flow within the Dviete floodplain at the beginning of the spring floods as mentioned above (Zavickis, Gruberts 1986).

In contrast, during the summer low water period the character of the water level fluctuation in the Dviete River differs considerably from that of the Daugava River at Vaikuļāni (Fig. 4). It highlights the importance of local weather conditions in the regulation of the surface water levels in the Dviete floodplain area during this hydrological phase. Large differences could also be observed when the absolute water levels at the Sloboda Bridge and the Daugava at Vaikuļāni are compared to each other for the time period between 2011 and 2013. In this case, the above mentioned differences in the character of the water level fluctuation at both sites are even more pronounced, especially when the summer–autumn low water period is analysed (Fig. 5).

When the results of the surface water level monitoring at the Dviete Bridge are compared to those obtained on the Daugava River at Vaikuļāni in 2005–2009, a linear correlation could be found, especially above the

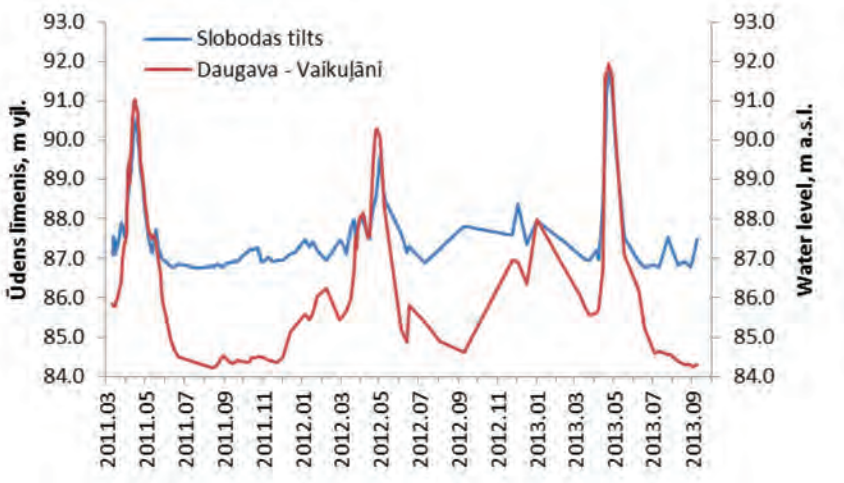

Figure 5. *Water level variation at the Dviete floodplain and the Daugava River at Vaikuļāni, 2011–2013 (after unpubl. data of Ā. Gruberte and the Latvian Environment, Geology and Meteorology Centre)*

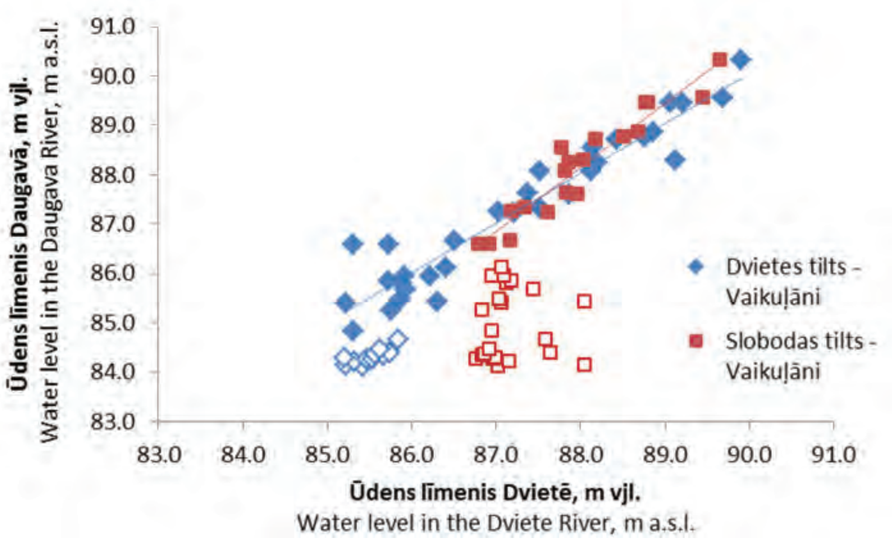

Figure 6. *Correlation between the water levels at the Dviete River and the Daugava River at Vaikuļāni, 2005–2009 (after unpubl. data of the Department of Geography, Daugavpils University, and the Latvian Environment, Geology and Meteorology Centre)*

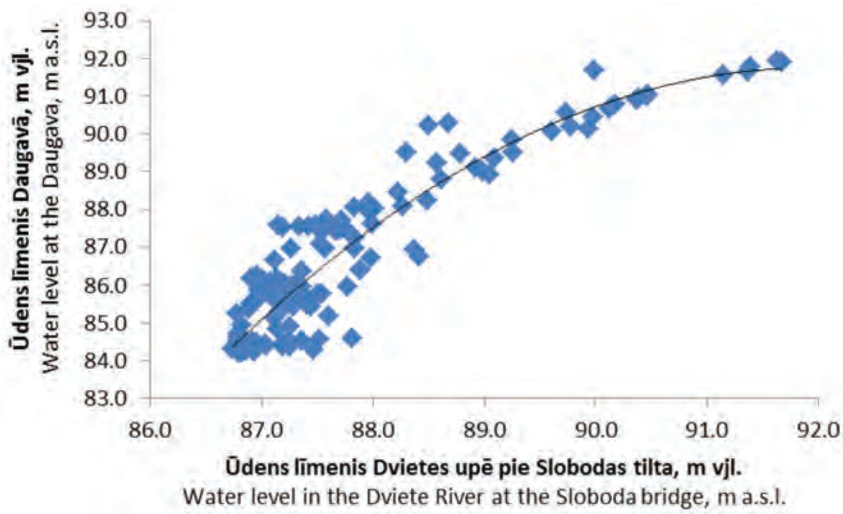

Figure 7. *Correlations between water levels at the Dviete floodplain and the Daugava River at Vaikuļāni, 2011–2013 (after unpubl. data of Ā. Gruberte and Latvian Environment, Geology and Meteorology Centre)*

absolute water level height of 85.0 m a.s.l. (Fig. 6). In the case of the Sloboda Bridge, the linear correlation exists starting from the absolute water level height of 86.5 m a.s.l. At lower water level heights there are no linear correlations stated, similarly to when the time period between 2011 and 2013 is also analysed (Fig. 7). In this case, a non-linear correlation exists between the water levels in the Daugava River at Vaikuļāni and the Dviete River at Sloboda Bridge. It is obviously related to the impact of local weather conditions and local hydrology during the summer and winter low water periods.

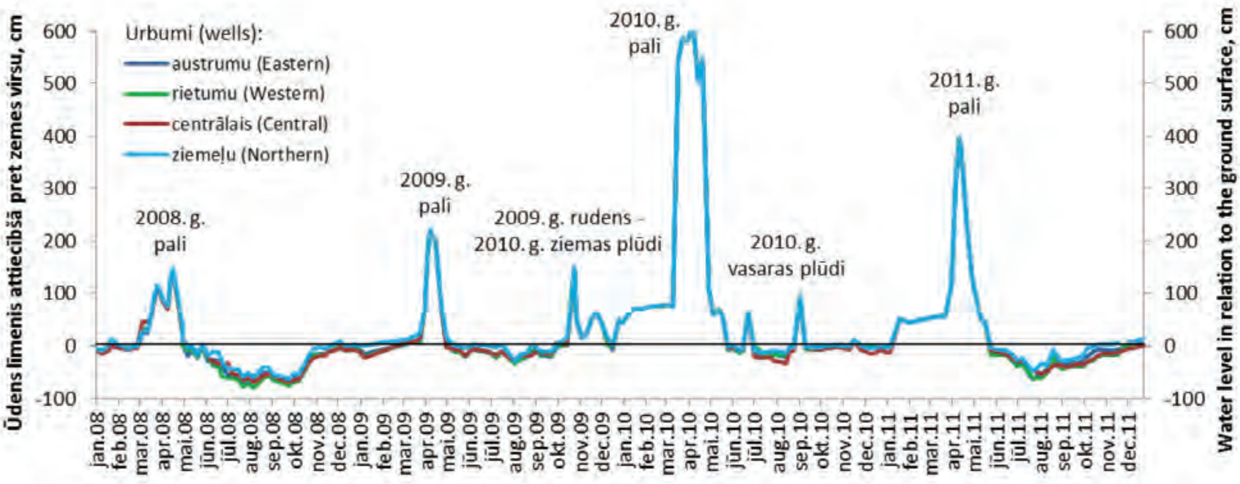

Figure 8. *Changes in the relative water level at the old monitoring wells, 2007–2011 (after A. Kriškijāne unpubl. data)*

Results of the groundwater level monitoring, 2008–2011

According to the groundwater level observations performed by Aija Kriškijāne at the old monitoring wells until 2012, the lowest groundwater levels in this section of the Dviete floodplain area could be observed during the summer low water period. Usually it starts in May or June. During these months, the water levels in the Dviete floodplain are rarely above the ground surface (Fig. 8). In contrast, the highest water levels of the year are related to an intrusion of the floodwaters from the Daugava River in March or April.

There were no significant differences observed between the groundwater levels in the four old monitoring wells during this period of monitoring, probably because they are located very close to each other, in similar hydrological conditions. It is necessary to highlight only the 'Northern' well, which is located closer to the old Dviete River meander and, therefore, has higher relative water levels, which are usually just below the ground surface (Fig. 8).

Precipitation was the most significant weather parameter that influenced the groundwater levels in the old monitoring wells during the summer low water periods. According to statistical analysis of decadal precipitation amounts recorded at the weather station „Putnusala" in 2008–2011, there was especially rainy weather in the 2nd decade of August and 1st decade of October, 2009. In 2010, the 3rd decade of June was very rainy, as was the 1st and 2nd decades of August and the 1st decade of September. In 2011, the same was also true for the 2nd decade of May and the 2nd decade in June (Fig. 9). In all these cases, the amount of precipitation reached or exceeded 40 mm per decade. However, there is no obvious relation of the groundwater level heights recorded at the monitoring wells to these precipitation records. Only in one case was the short-term flooding of the monitoring wells during the 3rd decade of June 2010 caused by a record-high amount of precipitation (about 60 mm per decade) (Fig. 8, 9). Actually, the groundwater levels rising to the ground surface was observed in the old monitoring wells even at relatively low decadal sums of precipitation (about 15–20 mm per decade). However, it was true only if such an intensity of precipitation was maintained for several weeks. This situation was observed, for example, in autumn 2008 and at the beginning of winter 2011/2012.

Precipitation was not recorded in April–June, 2009, due to technical problems.

During this observation period, the lowest groundwater levels in the old monitoring wells were recorded at the end of July–beginning of August, 2008 (about 60-80 cm below the ground surface) (Fig. 8). The lowest level was recorded in the 'Western' well on August 10, 2008 (-79.6 cm). Such record-low groundwater levels are obviously related to the two and a half month long meteorological drought period (from the 3rd decade of April to the 1st decade of July) when the total amount of precipitation in the Dviete floodplain area only reached 15 mm (Fig. 9). Very low groundwater levels were also recorded in September–October 2008 and July–August 2011, obviously related to prolonged drought periods with small amounts of precipitation.

In contrast to the summer low water period when the groundwater levels in the Dviete floodplain area are controlled by precipitation intensity, in wintertime they also depend on the air temperature. When it rises above 0o C, an infiltration of snowmelt waters is initiated to the floodplain soil. During this observation period, this situation occurred many times. For example, winter thaws were recorded in January–March 2008, November 2009, and February 2011. Therefore, the groundwater levels in wintertime were

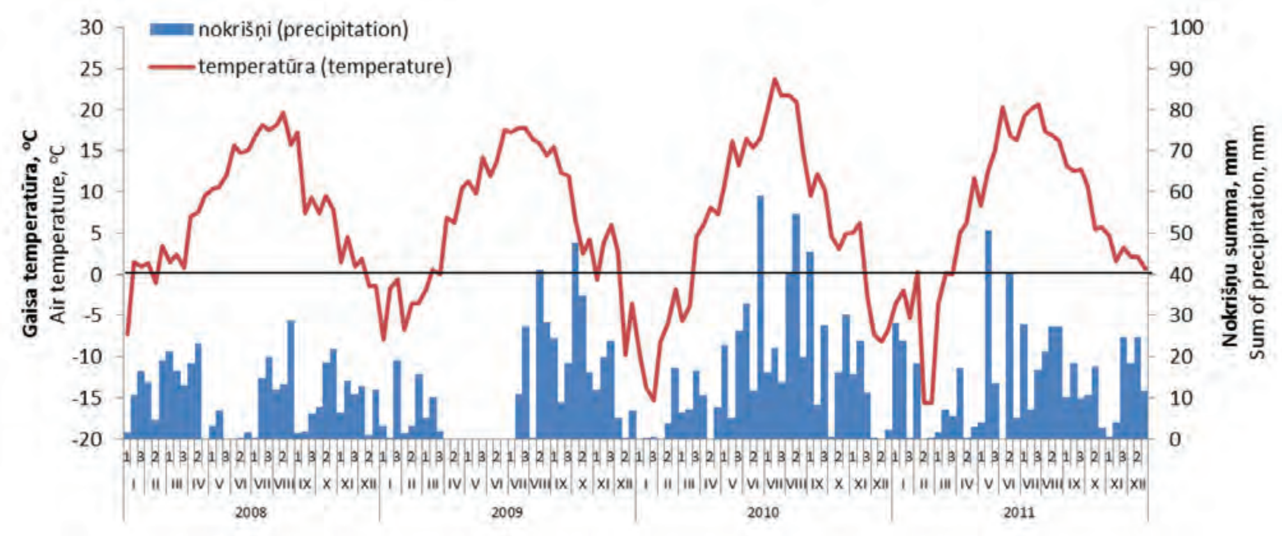

Figure 9. *Decadal variation of the average air temperature and sum of precipitation at the Dviete floodplain, 2008–2011 (data from the DU weather station „Putnusala").*

usually higher than those observed during the summer low water periods. Even more, the old monitoring wells were permanently flooded during the winters of 2009/2010 and 2010/2011 (Fig. 8).

Results of the groundwater level monitoring, 2012–2015

Observations of the groundwater levels in the new monitoring wells in 2012–2015 indicate that their relative heights fluctuate synchronously (Fig. 12, 13), and that they are closely related to the surface water hydrology and weather conditions in the Dviete floodplain area. During the summer low water periods (from May till October), the most different ones were the groundwater levels observed in monitoring wells N1 and N2. This can be explained by their location near the natural border of the floodplain, in a relatively high and dry place (Fig. 2, 3, 16). In the other wells, the short-term fluctuation of the groundwater levels was closely related to short-term changes in the amount of precipitation (Fig. 10). Each 10-15 mm of rain per week resulted in an increase of the groundwater levels by about 10 cm (Fig. 13, 15). In contrast, when the amount of precipitation was less than 10–15 mm per week, a marked decrease of the groundwater levels was observed.

During the winter low water periods the thaws were also very important because they refilled the groundwater horizons with snowmelt waters. For example, during the winter low water period of 2012/2013, an increase in the amount of the snowmelt water, as well as the height of the groundwater levels was observed when the air temperature reached 0°C (Fig. 11). In contrast, when the daily mean air temperatures dropped below -10°C in mid-January, the infiltration of snowmelt water to the groundwater horizons stopped. As a result, a quick fall in groundwater levels was observed. For example, at the monitoring well N1 the water level dropped by 0.5 m in one week (Fig. 11).

Regular observations of the groundwater levels in the old monitoring wells in 2012–2015 also point to the same influencing factors discussed above. Namely, the most remarkable decrease of the groundwater levels in these wells was observed during the summer months (July, August), under relatively dry weather conditions (Fig. 12, 13). During this observation period, the lowest relative groundwater level height (-87.0 cm) was recorded in the 'Western' well on August 23, 2015. On the other hand, a significant rise of the groundwater levels during the autumn months (September, October) was obviously related to an increase in the precipitation intensity (i.e. above 10 mm per decade for several decades in turn). The record-high amount of precipitation (more than 60 mm per decade) even caused a short-term flooding of the wells in July 2013 (Fig. 12–13).

Similarly, the impact of the winter thaws on groundwater levels was also observed during this observation period (for example, in January 2012 and the winter months of 2014/2015). During the last two years, local hydrological and weather conditions became even more important since the monitoring sites were not flooded by the Daugava's floodwaters in April and May as usual. As a result, during these months the water levels in the Dviete floodplain were very close to the ground surface and were regulated by local floods and weather conditions (Fig 12). In contrast, during the record-dry August in 2015, they dropped to the lowest levels ever observed.

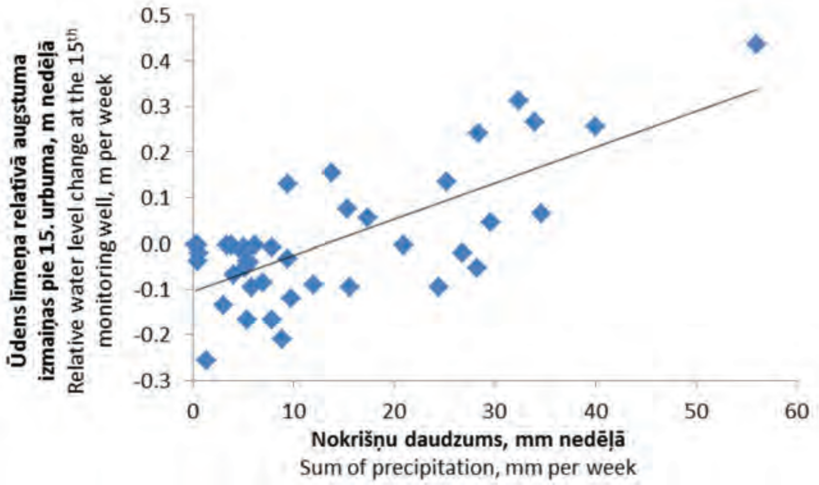

Figure 10. *Correlation between weekly precipitation records and changes in the relative height of the groundwater level at the Dviete floodplain during the summer-autumn low water periods, 2012–2013 (after unpubl. data of the Department of Geography, Daugavpils University)*

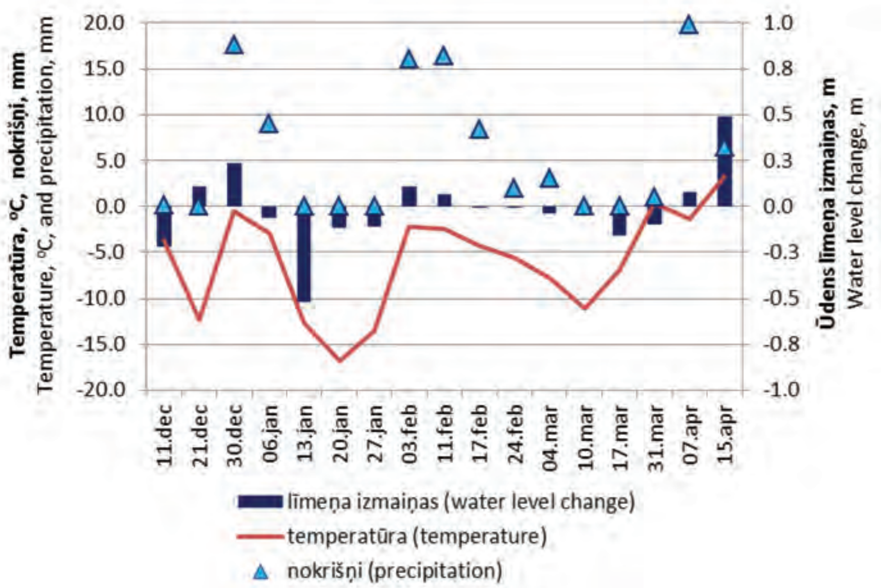

Figure 11. *Correlation of the groundwater level changes in the 1st monitoring well to variation trends in the air temperature and the snowmelt water supply in the Dviete floodplain, winter 2012/2013 (after unpubl. data of the Department of Geography, Daugavpils University)*

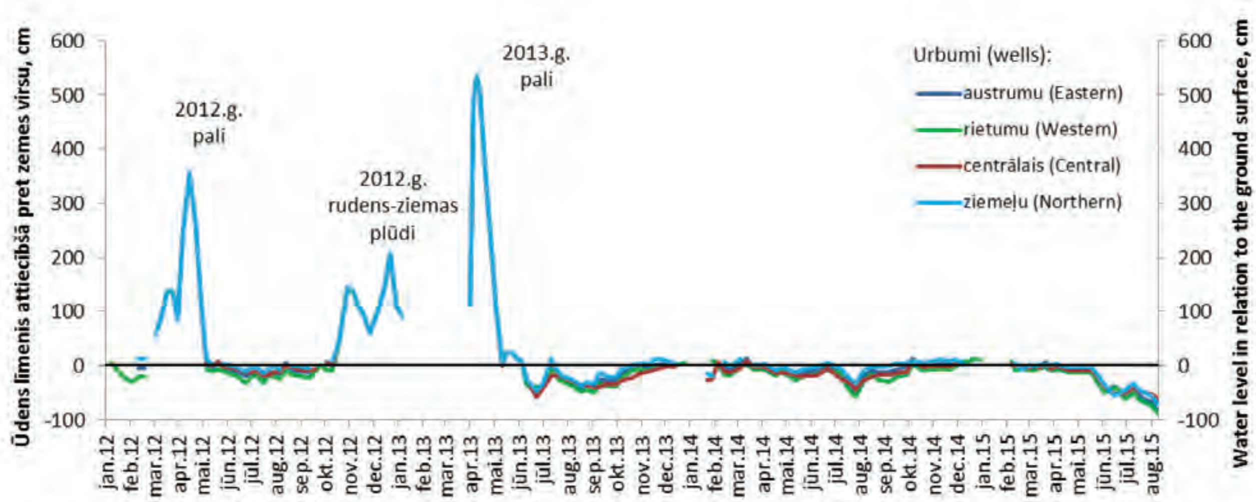

Figure 12. *Changes of the relative water level at the old monitoring wells, 2012–2015 (after observations made by the author)*

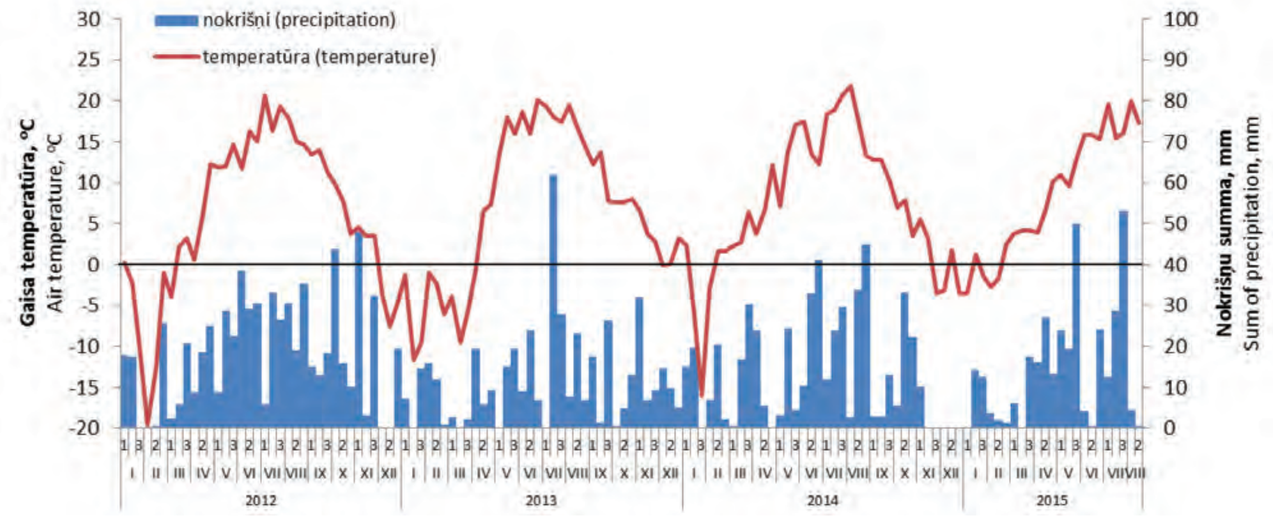

Figure 13. *Decadal variation of the average air temperature and sum of precipitation at the Dviete floodplain, 2012–2015 (data from the DU weather station „Putnusala"). Precipitation was not recorded in November–December, 2015, due to technical problems.*

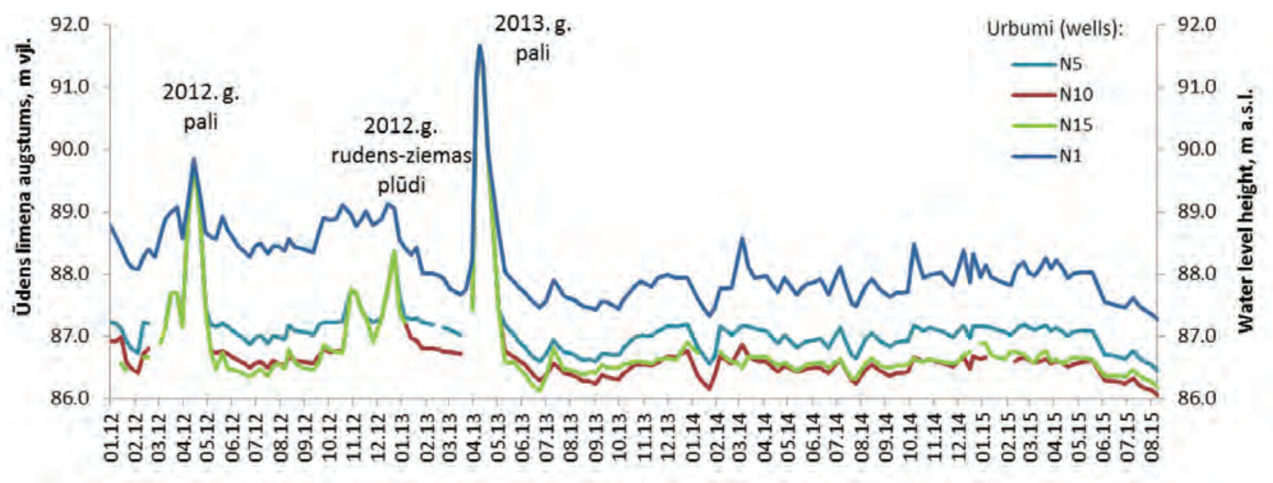

Figure 14. *Changes of the absolute water level at the new monitoring wells, 2012–2015 (after observations made by the author)*

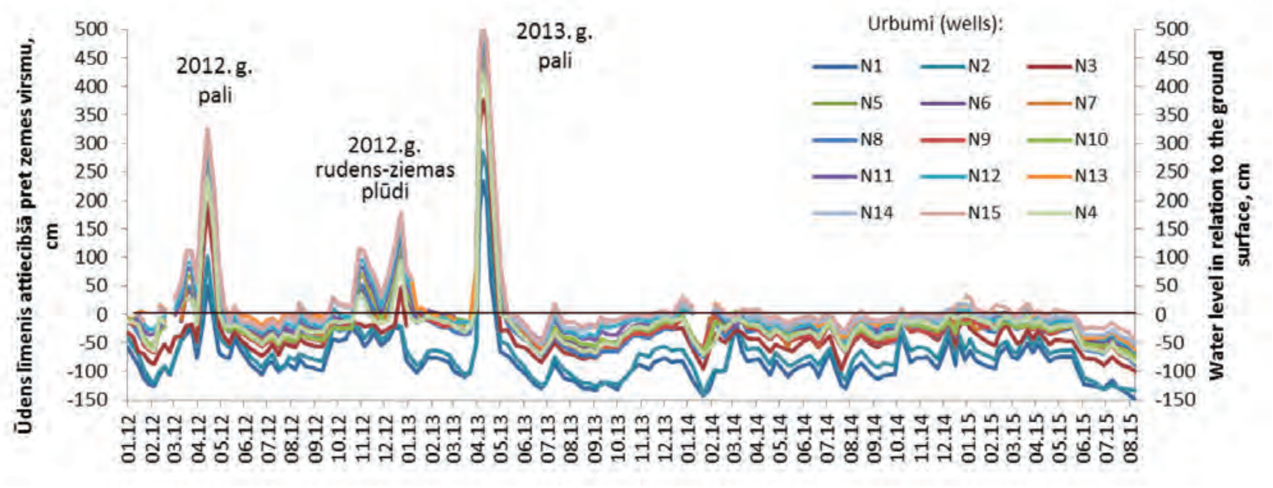

Figure 15. *Changes of the relative water level at the new monitoring wells, 2012–2015 (after observations made by the author)*

Due to a prolonged cold spell in the Dviete floodplain area in January and February 2012, a continuous decrease of the groundwater levels, formation of an ice cover and freezing of the monitoring wells was observed. It was interrupted by a winter thaw that started in mid-February (Fig. 13). At the end of March, under even higher main daily air temperatures, the ice cover in the wells melted away and the groundwater levels started to rise again (Fig. 14–15).

During the 1st decade of April they were flooded by the Daugava's floodwaters, and the absolute height of the surface water level reached 87.7 m a.s.l. This was followed by a short-term drop which was replaced by a second flood wave from the Daugava basin that flooded a large part of the Dviete floodplain area in mid-April. The highest water level in 2012 was recorded on April 29 (89.8 m a.s.l.). At this time, all the new monitoring wells were completely flooded. In contrast, the lowest water level in 2012 was recorded on July 9 (i.e. 86.3 m a.s.l. at the well N15).

After the spring floods of 2012, the groundwater levels repeatedly rose and fell in all the new monitoring wells in relation to weather conditions. These fluctuations correlated very well to the precipitation intensity variation in the Dviete floodplain area (Fig. 13, 15). The summer and autumn seasons of 2012 were quite rainy, with more than 30 mm of rain in some decades (Fig. 13). As a result, the groundwater levels were very close to the ground surface. A significant rise of the groundwater levels started in October, whereas in early November most of them were flooded once more (Fig. 15). At the beginning of December 2012 the air temperatures dropped below 0°C, and the water levels in the Dviete floodplain started to fall again. At the end of December a second flood wave followed, which reached its culmination in January 2013 (Fig. 14).

In January 2013, when negative air temperatures were finally established, a massive ice cover was formed within the Dviete floodplain that covered 13 of the 15 new monitoring wells. As the negative air temperatures persisted, the surface water levels in the floodplain meadows dropped, and the ice cover settled in. As a result, the plastic tubes of most of the monitoring wells were deformed or pushed deeper into the ground. Monitoring wells N7 and N14 were among the most damaged ones. Therefore, it was necessary to measure the relative heights of the wells' tubes above the ground surface once more in order to be able to compare the new groundwater level records to those obtained earlier.

Such measurements were performed in May and September, 2013, and they revealed that some tubes were pushed into the ground by 10–15 cm (for example, wells N7, N12, N13 and N14). However, the relative heights of some other tubes even increased when compared to the results obtained before (for example, by 24 cm at the well N3). Such a relative increase could be explained by the compaction of the soft floodplain soil profiles caused by the impact of the grazing animals. Settling of the soil surface near the new monitoring wells by about 3 cm was also stated during summer 2013. In this case, the compaction of the soil could be related to drying of the organic material.

Similarly to 2012, the groundwater levels in all the new monitoring wells fluctuated synchronously in 2013, according to local weather conditions and changes in the water levels in the Daugava River and the Dviete floodplain. And again, the most pronounced changes were observed during the spring floods (Fig. 15). In 2013, the spring floods started unusually late (in mid-April). It was the main factor that determined the record-high floodwater level and the very large size of the inundated area. The highest floodwater levels in the Daugava River at Vaikuļāni and the Dviete floodplain were recorded on April 23–24, i.e. 8–10 days later than usual. At the Sloboda Bridge, the highest floodwater level with an absolute height at 91.67 m a.s.l. was recorded on April 23. Although it was still 1.8 m below the absolute record of the 20th century observed in 1931, it was high enough to inundate all the monitoring wells. The 4.5 m high surface water measurement lath located at monitoring well N4 was also completely under water. Statistical analysis of the historic floodwater level records of the Daugava River at Vaikuļāni showed that the spring floods of 2013 were the highest ones since those observed in 1970 (Анон 1987).

After the peak of these record-high spring floods, the water levels gradually decreased both in the Daugava River and the Dviete floodplain. The floodwaters continued to drain back to the Daugava and the surface water levels continued to fall in all of May and June. The highest monitoring wells were only free of flooding in mid-May, while the others were still under water (Fig. 15, 16). At the end of May the floodwaters finally retreated to the Dviete River channel, and the summer low water period started.

The summer season of 2013 in the Dviete floodplain was relatively dry (Fig. 13). In July the groundwater levels dropped to the lowest marks recorded up to then (Fig. 14). The lowest groundwater level in 2013 was recorded on July 8 at monitoring well N15. It was at the absolute height of 86.10 m a.s.l. or -48 cm below the ground surface (Fig. 15).

In the second half of July, when the amount of precipitation reached 60 mm per decade, the groundwater levels in the new monitoring wells rose by 0.5–0.7 m (Fig. 12.) It was followed by a groundwater level decrease at the end of July – beginning of August, which continued until mid-September. In the second half of September the groundwater levels in all the wells rose once more, while in October they dropped

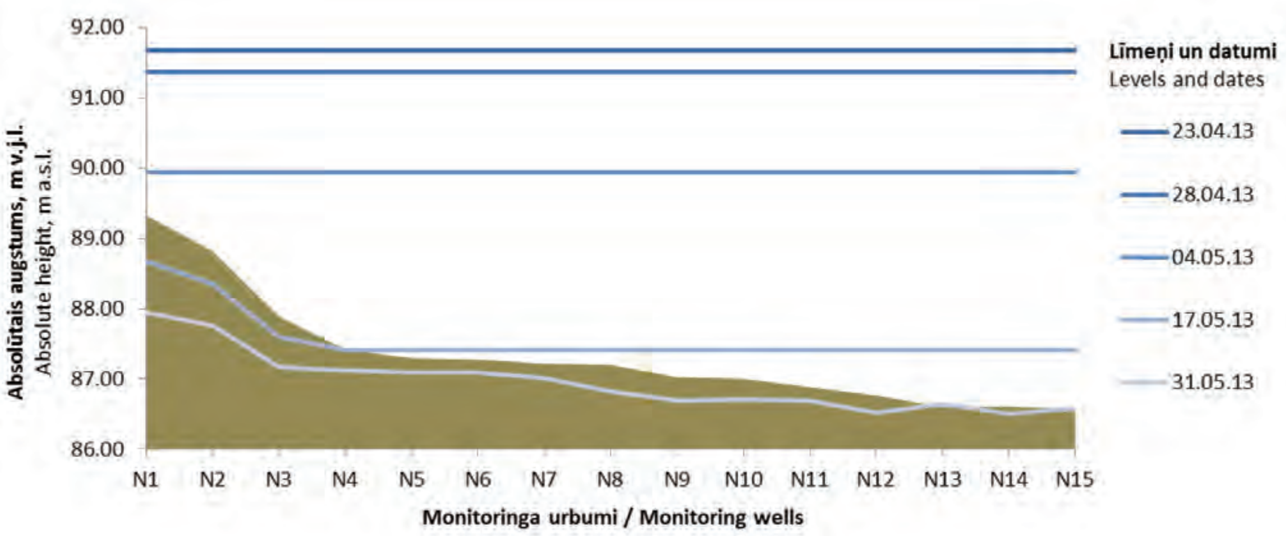

Figure 16. *Water level change in the Dviete floodplain during the drainage phase of the spring floods in 2013 (according to observations made by the author and Ā. Gruberte)*

back due to short-term changes in the amount of precipitation (Fig. 13, 15).

In general, the groundwater levels in 2013 were a little bit lower than those recorded during the summer low water period of 2012. They only returned to 'normal' levels at the end of 2013 when a sustained rainy period with positive air temperatures was established.

The beginning of winter 2014 was unusually warm and rich in thaws. Continuous cold weather persisted for only one month – from the 2nd decade of January till the 1st decade of February (Fig. 13). However, it was long enough for the groundwater levels to fall to new record-low levels for the winter low water period. The lowest groundwater levels were recorded on February 6 (Fig. 14, 15). The lowest relative level (-140 cm below the ground surface) was recorded at monitoring well N1, whereas the lowest absolute level (86.15 m a.s.l.) – at well N10. It's possible that these were not the absolute minimum records for the winter low water period here, and the levels were even lower closer to the Dviete River. Unfortunately, it was not possible to measure them precisely in all the new monitoring wells since some of them were completely frozen.

In mid-February, the first thaw of winter 2014 was recorded, and the groundwater levels in the new monitoring wells rose again by 40–50 cm in two weeks. After a short stabilisation of the water level in mid-March, the spring floods of 2014 started. These floods were unusually low when compared to those observed in other years. The highest water level of 2014 was recorded on March 28 (88.6 m a.s.l.). At this moment, the floodwaters from the Dviete River inundated only 4 out of 15 new monitoring wells. These were the first spring floods during this study when the water levels were below the ground surface in most monitoring wells.

The spring floods of 2014 were also very short – they continued for only two or three weeks. At the beginning of April the water level was already below the ground surface in all new monitoring wells except for well N15. During the prolonged summer low water period of 2014, the groundwater levels gradually fell. The lowest levels of 2014 were recorded on August 10 (Fig. 14, 15). During the next two decades 70 mm of rainwater were recorded in the Dviete floodplain area, and the groundwater levels rose again. As a result, monitoring well N15 was flooded again.

In the second half of October 2014 flash floods caused by rainstorms started in the Dviete floodplain area. At their culmination the water level in the Dviete floodplain reached almost the same absolute height (88.5 m a.s.l.) as in March 2014 (Fig. 14). Like in March, most monitoring wells were not flooded at all although the water levels were very close to the ground surface (Fig. 15). After these short floods the groundwater levels dropped again. However, the lowest possible winter levels were not reached even in December because the beginning of winter 2014/2015 was also unusually warm, with the mean decadal air temperature above -5°C (Fig. 13). Therefore, 2014 was characterised by very small annual amplitude of water level fluctuation and, also, by record-low groundwater levels at its beginning (see above).

Because of the winter thaws, there were winter floods and also a partial inundation of the monitoring wells at the beginning of 2015. On January 20, the highest water level in 2015 was recorded (88.3 m a.s.l.). In February, cold, dry weather was established in the

Dviete floodplain area, the floodwaters retreated and the groundwater levels dropped again. A small decrease in the groundwater levels this month could also be related to the demolition of beaver dams and the cleaning of the old Dviete River meanders further downstream.

During the 1st decade of March the mean daily air temperature rose above 0° C again, while during the 3rd decade of March an intense rainy period started (about 15 mm per decade). These processes resulted in a new rise in groundwater levels (Fig. 15). During March 2015, important changes also occurred in the hydrography of the Dviete River upstream from Skuķu Lake as its old meanders were cleaned and the straightened channel was blocked. As a result, the groundwater levels in the lower monitoring wells possibly changed, too.

The summer low water period of 2015 was not very rainy. Instead, there was a 'heat wave' recorded at the beginning of August (Fig. 13). As a result, the groundwater levels in all monitoring wells dropped significantly. In mid-August, the groundwater levels in most of the monitoring wells were as low as those recorded in February 2014 (Fig. 14, 15). Thus, new minimum records were stated for the summer low water period. The groundwater levels also continued to decrease during the second half of August. On August 23, 2015, new minimum records were stated for 18 out of the 19 monitoring wells used for this study. The lowest absolute groundwater level ever recorded (86.05 m a.s.l.) was observed at monitoring wells N10 and N14 (Fig. 14). The lowest relative height of groundwater level was recorded at well N1. It was -149 cm below the ground surface (Fig. 15).

This points to the necessity to restore not only the remaining old meanders of the Dviete River but also to restore the summer water level in the Skuķu Lake up to its natural height recorded before the drainage campaign of the Dviete floodplain during the 1930-ties. This problem has been investigated and discussed before (Račinskis 2005; Indriksons 2008; Uļjans, Gruberts 2010; Uļjans 2011), and the latest results of hydrological monitoring in the Dviete floodplain clearly demonstrates that it is necessary to solve it in the near future.

Conclusions

1. According to the results of surface water monitoring conducted in the Dviete floodplain until now, the highest water levels of the year are controlled by the hydrological regime of the Daugava. A strong linear correlation exists between water levels in the Dviete and Daugava rivers, especially starting from the absolute level 85.0 m a.s.l. at the Sloboda Bridge and 85.6 m a.s.l. at the Dviete Bridge. In such conditions, an increase of the water level in the Daugava River at Vaikuļāni results in a proportional increase of the water level in the Dviete floodplain.

2. The critical role of Daugava's hydrology in the regulation of the surface water level fluctuation in the Dviete floodplain is confirmed by the negative inclination of the floodwater surface between the Daugava, the Dviete Bridge and the Sloboda Bridge that forms at the peak of the spring floods and is oriented in the opposite direction to the usual water flow in the Dviete River. It facilitates the filling of the Dviete floodplain by Daugava's flood water masses.

3. Analysis of the results of groundwater level measurements conducted within the Dviete floodplain area from 2008–2011 shows that there is a synchronous fluctuation of the relative heights of the groundwater in all monitoring wells. It depends mainly on infiltration of the surface waters during the spring floods, the summer-autumn floods and the winter thaws. It also depends on variation in precipitation records in the Dviete floodplain area, especially during the summer-autumn low water period.

4. According to the results of monitoring of the groundwater levels at the new monitoring wells performed from 2012–2015, the height of the groundwater levels during the summer low water period depends on precipitation intensity. When it exceeds 15 mm per week the groundwater levels in monitoring wells start to rise, especially at those wells that are located closer to the river channel. In contrast, during the winter low water period air temperatures are more important. They control runoff of the snow melting waters to the Dviete River during winter thaws. Therefore, the air temperatures have an impact on the groundwater level fluctuation in the Dviete floodplain area as well.

5. Monitoring of the groundwater levels in the Dviete floodplain area encountered serious technical difficulties related to the flooding of the wells during the spring floods and repeated flooding in other seasons, as well as because of the formation of an ice cover at relatively high water levels in winter, its later compaction, and the deformation of the wells' tubes under its weight. Therefore, it is necessary to monitor the physical condition of the wells' tubes regularly, and to perform repeated measurements of their relative heights above the ground surface during the winter low water periods and, also, after the spring floods.

6. It is also necessary to restore the summer water level in the Skuķu Lake in the near future in order to manage the drying of the floodplain soils upstream from Skuķu Lake in dry summers even more effectively.

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References

Anon. 1987. Daugavpils rajona Dvietes zemienes inženieraizsardzības un lielmeliorācijas būvdarbu tehniski-ekonomiskais aprēķins. Rīga, Latvijas Valsts meliorācijas projektēšanas institūts, 65 lpp.

Gruberts D. 2003. The four largest floodplain lakes in Latvia: hydrology, hydrochemistry and hydrobiology. In: Järvet A., Lode E. (eds.) Ecohydrological Processes in Northern Wetlands. Selected Papers of International Conference and Educational Workshop. Tartu University Press, Tallinn, Estonia, 196–202.

Gruberts D. 2004. Daugavas palu pulss jeb atbūda Dvietes lukstos. – Vides vēstis, 3, 36.–39. lpp.

Gruberts D. 2006. Palu pulsa koncepcija Daugavas vidusteces palieņu ezeru ekoloģijā. Promocijas darbs. Daugavpils Universitāte, Ģeogrāfijas un ķīmijas katedra, Daugavpils, 152 lpp.

Gruberts D. 2007. Effect of floods on phytoplankton communities in aspect of river monitoring: a case of the Middle Daugava River (South-east Latvia). – Arch. Hydrobiol. Suppl., 161/3-4: 487–510.

Gruberts D. 2013a. Daugavpils Universitātes meteoroloģisko novērojumu stacija „Putnusala". Gadagrāmata. IV sējums, 2008. gads. Daugavpils, Daugavpils Universitātes akadēmiskais apgāds „Saule", 48 lpp.

Gruberts D. 2013b. Daugavpils Universitātes meteoroloģisko novērojumu stacija „Putnusala". Gadagrāmata. V sējums, 2009. gads. Daugavpils, Daugavpils Universitātes akadēmiskais apgāds „Saule", 48 lpp.

Gruberts D. 2013c. Daugavpils Universitātes meteoroloģisko novērojumu stacija „Putnusala". Gadagrāmata. VI sējums, 2010. gads. Daugavpils, Daugavpils Universitātes akadēmiskais apgāds „Saule", 48 lpp.

Gruberts D. 2013d. Daugavpils Universitātes meteoroloģisko novērojumu stacija „Putnusala". Gadagrāmata. VII sējums, 2011. gads. Daugavpils, Daugavpils Universitātes akadēmiskais apgāds „Saule", 48 lpp.

Gruberts D. 2013e. Daugavpils Universitātes meteoroloģisko novērojumu stacija „Putnusala". Gadagrāmata. VIII sējums, 2012. gads. Daugavpils, Daugavpils Universitātes akadēmiskais apgāds „Saule", 48 lpp.

Gruberts D. 2014a. Daugavpils Universitātes meteoroloģisko novērojumu stacija „Putnusala". Gadagrāmata. IX sējums, 2013. gads. Daugavpils, Daugavpils Universitātes akadēmiskais apgāds „Saule", 48 lpp.

Gruberts D. 2014b. Dvietes palienes virszemes un pazemes ūdens līmeņa novērojumu rezultātu izvērtējums LIFE+ projekta „Dviete" kontekstā. Daugavpils Universitātes 56. starptautiskā zinātniskā konferences tēzes. Daugavpils Universitātes akadēmiskais apgāds „Saule", 58. lpp.

Gruberts D. 2015. Daugavpils Universitātes meteoroloģisko novērojumu stacija „Putnusala". Gadagrāmata. X sējums, 2014. gads. Daugavpils, Daugavpils Universitātes akadēmiskais apgāds „Saule", 48 lpp.

Gruberts D., Druvietis I., Kļaviņš M. 2005. Seasonal variability of aquatic chemistry and phytoplankton communities in a shallow floodplain lake of the Daugava River, Latvia. – Ecohydrology & Hydrobiology, 5 (2),): 155–164.

Gruberts D., Druvietis I., Parele E., Paidere J., Poppels A., Prieditis J., Škute A. 2007. Impact of hydrology on aquatic communities of floodplain lakes along the Daugava River (Latvia). – Hydrobiologia, 584: 223–237.

Gruberts D., Soms J. 2004. Dvietes senlejas ģenēzes jautājumi. Krāj.: Ģeogrāfija, Ģeoloģija, Vides zinātne. LU 62. zin. konf. tēzes. LU Akadēmiskais apgāds, Rīga, 133–137. lpp.

Gruberts D., Vilcāne K. 2015. Floodwater storage capacity of the Middle Daugava floodplain. In: "Environment. Tehnology. Resources". Conference Proceedings. Rēzekne Higher Educational Institution, Rēzekne, 112–115.

Indriksons A. 2008. Dvietes upes meandru atjaunošanas hidroloģiskā priekšizpēte. Pārskats. Salaspils, Rīga: LVMI „Silava", Latvijas Ornitoloģijas biedrība. 67 lpp.

Indriksons A. 2010. Dvietes upes palienes hidroloģiskais režīms un dabiskās gultnes atjaunošanas projektēšana. Krāj.: Ģeogrāfija, Ģeoloģija, Vides zinātne. LU 68. zin. konf. tēzes. LU Akadēmiskais apgāds, Rīga, 89–90. lpp.

Paidere J. 2008. Zooplanktons Daugavas upes-palieņu sistēmā. Klimata mainība un ūdeņi. Rakstu krājums. Latvijas Universitāte, Rīga, 83–91. lpp.

Paidere J. 2012. Zooplanktona cenožu mainība Daugavas vidusteces upes-palienes sistēmā dažādās hidroloģiskā režīma fazēs. Promocijas darbs. Daugavpils Universitāte, Ekologijas institūts, Daugavpils, 112 lpp.

Paidere J., Gruberts D., Škute A., Druvietis I. 2007. Impact of two different flood pulses on planktonic communities of the largest floodplain lakes of the Daugava River (Latvia). – Hydrobiologia, 592: 303–314.

Račinskis E. (red.) 2005. Dabas parka "Dvietes paliene" dabas aizsardzības plāns 2006.-2016. gadam. Rīga, LOB, 73 lpp.

Stakle P., Kanaviņš E. 1941. Latvijas iekšzemes ūdeņu hidrometriskie pētījumi no 1929.g. 1.XI līdz 1940. g. 31.X. Jūrniecības pārvaldes izdevums, Rīga, 623 lpp.

Škute A., Gruberts D., Soms J., Paidere J. 2008. Ecological and hydrological functions of the biggest natural floodplain in Latvia. – Ecohydrology & Hydrobiology, 8 (2–4): 291–306.

Uļjans J. 2009. Ūdens sastāva izmaiņas Dvietes palienē dažādās hidroloģiskā režīma fāzēs. Bakalaura darbs. Daugavpils Universitāte, Ģeogrāfijas un ķīmijas katedra, Daugavpils, 36 lpp.

Uļjans J. 2011. Priekšizpēte un ieteikumi Skuķu ezera atjaunošanai. Maģistra darbs. Daugavpils Universitāte, Ģeogrāfijas un ķīmijas katedra, Daugavpils, 54 lpp.

Uļjans J., Gruberts D., 2010. Skuķu ezera atjaunošanas nepieciešamība. Krāj.: Ģeogrāfija. Ģeoloģija. Vides zinātne. Referātu tēžu krājums. Latvijas Universitātes 68. zinātniskā konference. Rīga, 2010.g. 03.februāris. Rīga, LU Akad. apgāds, 244.–245. lpp.

Zavickis J., Gruberts R. 1986. Tilts pār Dvieti autoceļā Bebrene-Zariņi-Dviete. Tehniskais projekts. A/S „Ceļuprojekts", Rīga, 14.–17. lpp.

Глазачева Л. И. 1965. Ледовой и термический режим рек и озер Латвийской ССР. Ученые записки, Географические науки. Том 65. Зваигзне, Рига, 232 стр.

Анон. 1987. Государственный водный кадастр, Том X. Латвийская ССР. Многолетние данные о режиме и ресурсах поверхносных вод суши. Ленинград, Гидрометеоиздат, 31, 276 стр.

Эберхард Г. Я. 1972. Строение и развитие долин бассейна реки Даугава. Зинатне, Рига, 21–394 стр.

Population change of the Corncrake *Crex crex* and other farmland bird species in the Dviete floodplain nature park from 2006 to 2015

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Summary

In 2010 the "Restoration of Corncrake habitats in Dviete floodplain Natura 2000 site" project (LIFE09 NAT/LV/000237; hereinafter – the project) was started by the Latvian Fund for Nature. In order to improve the habitat quality for the Corncrake Crex crex, from 2011 to 2015 bushes were cut and the existing pastures of 'Konik' horses and 'Highlander' cattle were expanded.

In order to evaluate the impact of the habitat restoration measures on the Corncrake and other nocturnal farmland bird species, monitoring of nocturnal farmland birds was started in the project site in 2011. To evaluate the long-term population trends for the entire Natura 2000 site data from counts since 2006 was also used. The counts were carried out in seven routes, two of which (with a total length of 10.2 km) represent the project site, but five (21.9 km) are used as a 'control' and represent Natura 2000 as a whole. The latter routes are the ones used for monitoring since 2006. Standard methods used in the national monitoring scheme of nocturnal birds were used with the only exception being the lack of habitat mapping.

During 2006–2015, a total of twenty-five species of nocturnal farmland birds have been recorded in the Dviete floodplain nature park. The Corncrake was recorded in all seven routes. The population trend of the Corncrake in the project area is increasing at a significantly higher level than in the nature park as a whole. The fluctuation in the population trend within the project area may be related to a delayed-in-season vegetation recovery of suitable conditions for the Corncrake after management activities.

Habitat selection by the Corncrake was also analysed in the project site in order to evaluate the impact of habitat management on Corncrake habitat quality. We were able to collect too little data to make definite conclusions. It was possible to test the significance of the selection only in the case of pastures, and it was found that selection of pastures was not significantly different from random selection.

A significant increase in the population trend of the Great Snipe *Gallinago media* is found in the project territory. This is opposite to the long term trend in the Dviete floodplain nature park as a whole and probably is related to management activities. Decreasing trends of the Spotted Crake *Porzana porzana*, Common Grasshopper Warbler *Locustella naevia* and Blyth`s Reed Warbler *Acrocephalus dumetorum* were detected, possibly as a result of bush removal.

Material and methods

The main aim of the project LIFE09 NAT/LV/000237 "Restoration of Corncrake habitats in Dviete floodplain Natura 2000 site" implemented by the Latvian Fund for Nature (further – the project), was to increase the quality of Corncrake Crex crex breeding habitats. To reach this aim, bushes were removed from 2011 to 2015, pastures of 'Konik' breed horses and cattle fitted to live outdoors all-year-round were expanded from 2011 to 2013 and stumps of trees and bushes were mulched to suppress the regrowth of shoots from 2014 to 2015.

To assess the management's influence on breeding bird populations, monitoring of the Corncrake, as well as the rest of nocturnal farmland bird species, was established in 2011. To assess the longer term bird population changes in the entire Dviete floodplain nature park, data from the previous project LIFE04NAT/

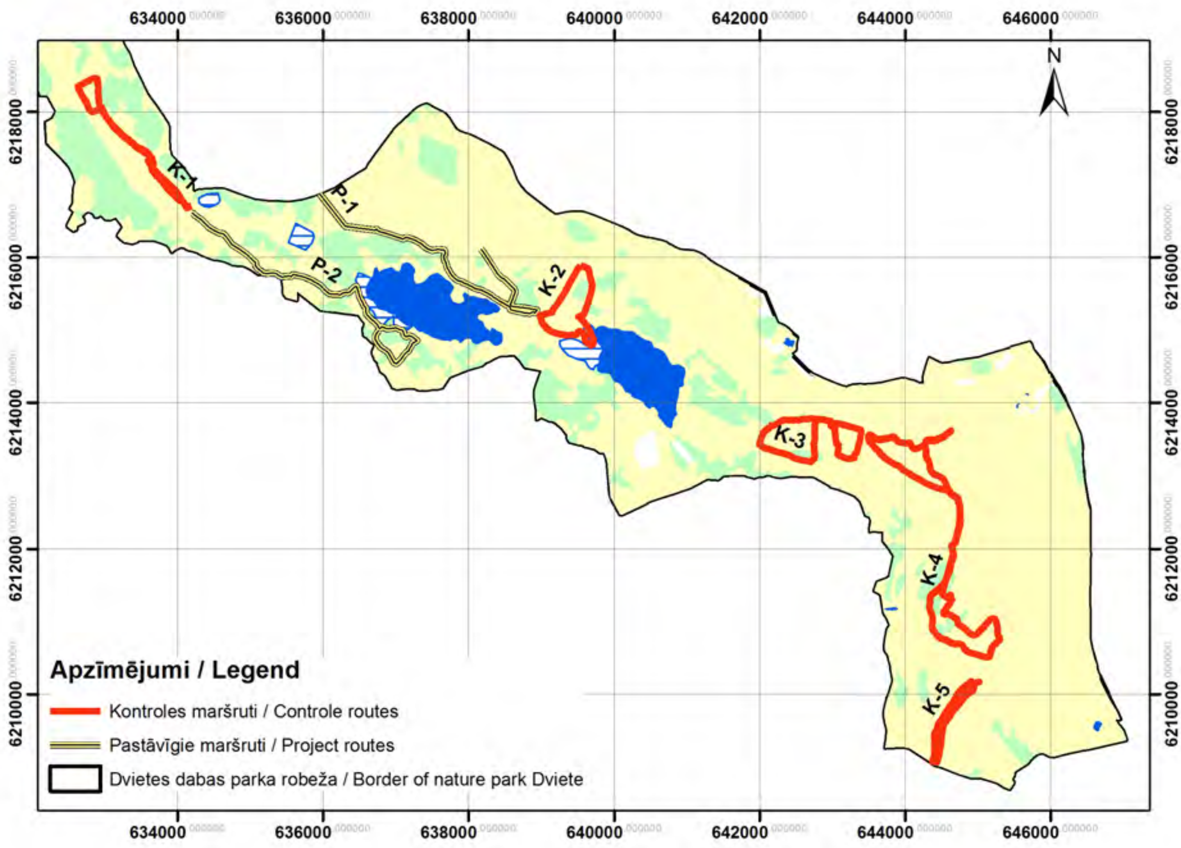


Figure 1. *Location of the routes for nocturnal bird counts: K-1–K-5 – 'control' routes, P-1 and P-2 – routes in the project area.*

Table 1
Length of the routes for nocturnal bird counts in the Dviete floodplain nature park

Route*	Length (km)
K-1	3,22
K-2	3,24
K-3	4,11
K-4	8,75
K-5	2,60
P-1	5,09
P-2	5,11

* See the routes in Fig. 1.

LV/000198 "Restoration of Latvian floodplains for EU priority species and habitats" (Račinska, Klepers 2008; years 2006–2008) and "Natura 2000 site monitoring. Birds" (unpubl. data of Latvian Ornithological Society, 2009) were used. Corncrake population change was visually compared to the trend from the nationally coordinated background monitoring scheme (Keišs 2014, years 2006–2014).

Two bird census routes (P-1 and P-2 in Fig. 1) were established to assess project management influences, for the census period 2011–2015. To study population change in the entire Dviete floodplain nature park, `control` routes from the previous project were used, for census period 2006–2009 and 2011–2015. The total length of the `control` routes was 21.9 km, the `project` routes – 10.2 km. Control routes in the entire period (2006–2009 and 2011–2015) were counted by J. Reihmanis, while in project route P-2 by A. Avotiņš jun. and P-1 by D. Drazdovskis (2011–2012 and 2014–2015) and by G. Grandāns (2013).

Table 2

Census dates in the Dviete floodplain nature park 2011-2015

Route*		K-1	K-2	K-3	K-4	K-5	P-1	P-2
2011. g.	1. uzsk. 2. uzsk.	27.05. 20.06.	27.05. 20.06.	26.05. 18.06.	27.05. 19.06.	07.06. 20.06.	01.06. 19.06.	01.06. 09.06.
2012. g.	1. uzsk. 2. uzsk.	06.06. 23.06.	06.06. 23.06.	05.06. 22.06.	06.06. 23.06.	07.06. 23.06.	24.05. 09.06.	09.06. 20.06.
2013. g.	1. uzsk. 2. uzsk.	09.06. 21.06.	07.06. 21.06.	08.06. 22.06.	06.06. 21.06.	09.06. 23.06.	06.06. 21.06.	06.06. 21.06.
2014. g.	1. uzsk. 2. uzsk.	10.06. 26.06.	10.06. 26.06.	09.06. 25.06.	10.06. 26.06.	11.06. 27.06.	03.06. 17.06.	06.06. 16.06.
2015. g.	1. uzsk. 2. uzsk.	09.06. 24.06.	08.06. 25.06.	08.06. 26.06.	09.06. 25.06.	10.06. 26.26.	05.06. 21.06.	05.06. 17.06.

* Census route nummeration as in Fig. 1

Bird censuses were done according to the national monitoring scheme "Nocturnal farmland bird monitoring" (Keišs 2006a). The only difference from the scheme mentioned is a lack of habitat mapping. In 2011 and 2012 project routes were counted three times and control routes twice. For the population estimate, randomly chosen census was selected for the project routes from 2011 and 2012. All data were used in Corncrake habitat selection analysis. Census dates gathered in Table 2.

All the species detectable at night were registered in the project routes, while in the control routes only the Corncrake *Crex crex*, Common Quail *Coturnix coturnix*, Spotted Crake *Porzana porzana*, Great Snipe *Gallinago media*, Common Grasshopper Warbler *Locustella naevia* and River Warbler *Locustella fluviatilis* were registered.

The bird population changes were analysed in groups – control routes from 2006 to 2015 and the project routes from 2011 to 2015, using MS Access tool BirdSTATs (van der Meij 2007) with base year 2011. The seasonal average per route was used in indexing.

There were three approaches used to estimate the influence of project management activities on nocturnal farmland bird population change and habitat quality, especially the Corncrake:

1. Comparison of population trends between control and project routes (for species registered in control routes, see before);
2. Population trends since the start of project management activities (for species not registered in control routes);
3. Corncrake habitat selection in relation to management activities

To establish the influence of management activities on Corncrake habitat quality, habitat selection was analysed using the index of choice – the forage ratio:

$w_i = o_i / \pi_i$, where

o_i – ratio of used resource units per stratum i;

π_i – ratio of available resources per stratum i (Manly *et al.* 2002).

In this case, strata was habitat classes, units used – all the Corncrake observation points, while the available resource unit proportion was a ratio of each habitat class per surveyed area. The surveyed territory size was estimated by creating a buffer around routes with a width of 460 m (the maximum distance of Corncrake observation from the route, except one outlying observation 660 m from the route). Only that part of the surveyed area which lies in the nature park covering 760.3 ha (Fig. 2) was analysed, but during analysis the landscape class "Lake" was eliminated leading to an area of 684.3 ha (Table 3 and Table 4).

For habitat categorisation the electronic land use map created by State Land Service of the Republic of Latvia was combined with the project management activities layer created by the Latvian Fund for Nature. Information on the habitat areas is collected in Table 3, but for practical reasons, for different analysis habitat classes were combined with project nature management activities used (Table 4).

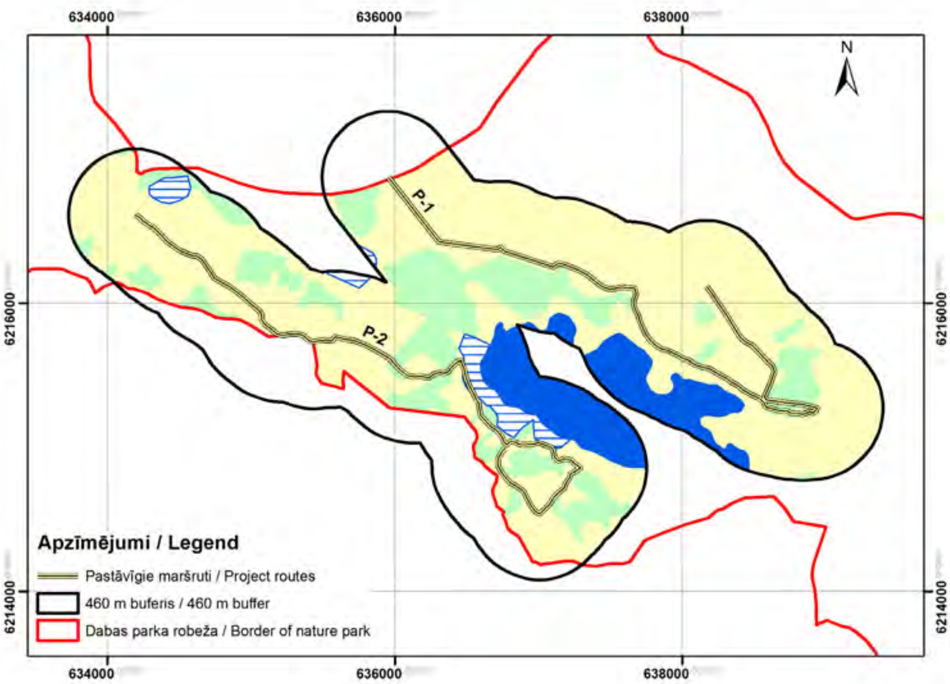


Figure 2. *Territory within 460 m of the routes that were considered surveyed when evaluating the habitat selection of Corncrake Crex crex.*

Table 3
Areas of habitat types in the surveyed territory of the project area (in 2015)

Habitat type *	Area (ha)
Passable mire	6.0
Passable mire, pasture	8.6
Passable mire, bushes cut	0.2
Passable mire, bushes cut, pasture	2.6
Passable mire, bushes cut, stumps mulched, pasture	3.0
Lake	69.5
Lake, pasture	1.7
Lake, bushes cut	5.9
Lake, bushes cut, pasture	4.0
Lake, bushes cut, stumps mulched	1.1
Lake, bushes cut, stumps mulched, pasture	0.2
Forest	65.0
Forest, pasture	40.8
Forest, bushes cut	0.4
Forest, bushes cut, pasture	10.8
Forest, bushes cut, stumps mulched	1.0
Forest, bushes cut, stumps mulched, pasture	28.0
Field	309.0
Field, pasture	165.3
Field, bushes cut	2.5
Field, bushes cut, pasture	14.7
Field, bushes cut, stumps mulched	4.6
Field, bushes cut, stumps mulched, pasture	15.5
Total	**760,297**

* "Passable mire", "Lake", "Forest" and "Field" are the land use categories according to the Satellite Map of Latvia, after the commas the managament measures used are indicated.

Table 4

*Merged habitat types used for the analysis of habitat selection by the Corncrake (in 2015)**

Habitat type	Area (ha)**
Bushes cut	94.4
Bushes not cut	665.9
Pasture	295.0
No pasture	465.3
Stumps mulched	53.3
Stumps not mulched	707.0
Pasture, bushes cut	32.2
Pasture, bushes not cut	216.4
No pasture, bushes cut	9.0
No pasture, bushes cut, stumps mulched	6.8
No pasture, bushes not cut, stumps not mulched	449.5
Pasture, bushes cut, stumps mulched	46.5

* All land-use categories (except "Lake" which is not used in the analysis) of the Satellite Map of Latvia are merged.

** Areas of different categories overlap.

The chi-squared (χ^2) test was used to evaluate observed Corncrake habitat selection as opposite to random selection.

To estimate the density of singing Corncrake males in different habitats, we used the previously mentioned habitat categories with the maximum number of Corncrake observations.

Results

During monitoring (2006–2015) 25 nocturnal bird species were observed (Table 5).

The Corncrake was observed in all the routes from 2011–2015 (Annex 1).

A large enough dataset to analyse population changes in the Dviete floodplain nature park was gathered for six species – the Spotted Crake, Corncrake, Snipe, Great Snipe, Common Grasshopper Warbler and River Warbler. Regarding to the project territory, population change could be analysed for nine more species – the Common Quail, Woodcock, Thrush Nightingale, Seivi`s Warbler, Sedge Warbler, Blyth`s Reed Warbler and River Warbler. Population change indices are gathered in Annex 2. Five of the species had significant changes in population trends in both in the nature park and the project territory (Table 6).

Table 5

Nocturnal birds registered in census routes in the Dviete floodplain nature park from 2006–2015

Species	
Bittern *Botaurus stellaris*	Long-Eared Owl *Asio otus*
Little bittern *Ixobrychus minutus*	Short-Eared Owl *Asio flammeus*
Black Grouse *Tetrao terix*	Thrush Nightingale *Luscinia luscinia*
Quail *Coturnix coturnix*	Whinchat *Saxicola rubetra*
Water Rail *Rallus aquaticus*	Common Grasshopper Warbler *Locustella naevia*
Spotter Crake *Porzana porzana*	River Warbler *Locustella fluviatilis*
Little Crake *Zapornia parva*	Saivi`s Warbler *Locustella luscinioides*
Corncrake *Crex crex*	Sedge Warbler *Acrocephalus schoenobaenus*
Lapwing *Vanellus vanellus*	Blyth`s Reed Warbler *Acrocephalus dumetorum*
Snipe *Gallinago gallinago*	Marsh Warbler *Acrocephalus palustris*
Great Snipe *Gallinago media*	Reed Warbler *Acrocephalus scirpaceus*
Woodcock *Scolopax rusticola*	Great Reed Warbler *Acrocephalus arundinaceus*
Tawny Owl *Strix aluco*	

Table 6

Bird species with significant population trend change, either in the Dviete floodplain nature park or in the project area

Species	Population trend	
	In the nature park (since 2006)	In the project area (since 2011)
Corncrake *Crex crex*	moderate increase	steep increase
Spotted Crake Porzana porzana	uncertain	steep decline
Great Snipe *Gallinago media*	moderate decline	steep increase
Common Grasshopper Warbler *Locustella naevia*	steep decline	uncertain
Blyth`s Reed Warbler Acrocephalus dumetorum	–	steep decline

It can be seen (Annex 2) that the population change in the nature park is similar to the national trend (Auniņš et al. 2014), while the trend calculated from project routes shows a rapid increase. The differences in trends and index fluctuations in the project area can be explained by nature management activities done in the project.

The analysis of Corncrake habitat selection in relation to bush removal shows that in 2012 and 2015 the Corncrakes preferred areas where bushes were cut and in 2013 and 2014 – areas where bushes were not cut (Table 7). Pastures were preferred in 2013 and 2015, while there were higher index values for areas without pasture in 2011, 2012 and 2014 (Table 8). With an increase in the mulched area, those sites were preferred in 2015, but in 2014, when the mulched area was just 2 % of that surveyed, there were no Corncrakes observed there (Table 9).

When analysing the effect of all the management activities on the Corncrake's preferred habitat, it can be seen that in 2014 (as well as in 2013) the Corncrake preferred pastures with no bushes removed (Table 10), while in 2012 the opposite was true – index values were higher in pastures where bushes were removed, and Corncrakes avoided pastures, where bushes were not removed. In 2015 forage ratio values were higher in pastures where bushes were removed, than in those pastures without bush removal.

In the rest of the cases the number of observed Corncrakes was less than 5, therefore the proper use of the chi-squared (χ^2) test was only possible for habitat selection related to pastures. In none of these years did the choice differ from random (2011: χ^2=1.44, n.s.; 2012: χ^2=1.51, n.s.; 2013: χ^2=0.78, n.s.; 2014: χ^2=0.54, n.s.; 2015: χ^2=1.35, n.s.).

In 2012 the largest densities of Corncrake were observed in sites with bushes removed and no pasture, followed by pastures with bushes removed. In 2013 – in pastures, in 2014 – territories with no management and in 2015 – pastures with bushes removed, followed by pastures with bushes removed and stumps mulched (Table 11).

Table 7

*Habitat selection of the Corncrake in the project area depending on bushes cut **

Habitat	2012			2013			2014			2015		
	π_i	o_i	w_i	π_i	o_i	w_i	π_i	o_i	w_i	π_i	o_i	w_i
Bushes removed	0,06	0,12	2,04	0,09	0,07	0,69	0,13	0,00	0,00	0,12	0,13	1,01
Bushes not removed	0,94	0,88	0,94	0,91	0,93	1,03	0,87	1,00	1,15	0,88	0,88	1,00

* π_i – population proportion; o_i – used sample proportion; w_i – forage ratio

Table 8

*Habitat selection of the Corncrake in the project area, depending on the establishment of a pasture**

Habitat	2011			2012			2013			2014			2015		
	π_i	o_i	w_i	π_i	o_i	w_i	π_i	o_i	w_i	π_i	o_i	w_i	π_i	o_i	w_i
Pasture	0,25	0,17	0,66	0,27	0,16	0,60	0,39	0,46	1,16	0,43	0,25	0,59	0,39	0,42	1,07
No pasture	0,75	0,83	1,12	0,73	0,84	1,15	0,61	0,54	0,90	0,57	0,75	1,31	0,61	0,58	0,95

* π_i – population proportion; o_i – used sample proportion; w_i – forage ratio

Table 9

*Habitat selections of the Corncrake in the project area depending on stump mulching**

Habitat	2011			2012			2013			2014			2015		
	π_i	o_i	w_i	π_i	o_i	w_i	π_i	o_i	w_i	π_i	o_i	w_i	π_i	o_i	w_i
Stumps mulched	–	–	–	–	–	–	–	–	–	0,02	0	0	0,07	0,08	1,19
No mulching	–	–	–	–	–	–	–	–	–	0,98	1,02	2,36	0,93	0,92	0,99

* π_i – population proportion; o_i – used sample proportion; w_i – forage ratio

Table 10

The influence of habitat management on habitat preference by the Corncrake

		Pasture, bushes, removed	Pasture, bushes not removed	No pasture bushes removed, not mulching	No pasture bushes removed, mulched	No pasture bushes not removed, not mulched	Pasture bushes removed, mulched
2011	π_i	–	0,25	–	–	0,75	–
	o_i	–	0,17	–	–	0,83	–
	w_i	–	0,66	–	–	1,12	–
2012	π_i	0,04	0,23	0,02	–	0,71	–
	o_i	0,04	0,12	0,08	–	0,76	–
	w_i	1,04	0,52	3,93	–	1,07	–
2013	π_i	0,07	0,32	0,02	–	0,59	–
	o_i	0,07	0,39	0,00	–	0,54	–
	w_i	0,88	1,23	0,00	–	0,93	–
2014	π_i	0,11	0,32	0,02	0,00	0,55	0,02
	o_i	0,00	0,25	0	0	0,75	0
	w_i	0,00	0,78	0	0	1,36	0
2015	π_i	0,04	0,28	0,01	0,01	0,59	0,06
	o_i	0,13	0,29	0,00	0,00	0,58	0,08
	w_i	2,96	1,02	0,00	0,00	0,99	1,36

* π_i – population proportion; o_i – used sample proportion; w_i – forage ratio

Table 11
Density of Corncrakes (males per 100 ha) in territories with various habitat management

Habitat	2011. g.	2012. g.	2013. g.	2014. g.	2015. g.
Pasture, bushes removed	–	3,8	5,9	0,0	9,3
Pasture, bushes not removed	2,5	0,6	5,5	1,8	3,2
No pasture, bushes removed, not mulched	–	7,2	0,0	0,0	0,0
No pasture, bushes removed, mulched	–	–	–	0,0	0,0
No pasture, bushes not removed, not mulched	3,5	1,6	2,7	1,36	3,1
Pasture, bushes removed, mulched	–	–	–	0,0	4,3
Total in project territory	**2,8**	**1,6**	**3,8**	**2,3**	**3,2**

Discussion

Corncrake monitoring data shows a statistically significant population increase in the Dviete floodplain nature park (moderate) as well as the project territory (steep). It is plausible that fluctuations in the project territory are related to vegetation recovery after certain management activities. Most probably factors from outside the nature park are also influencing population trends, because those from the nature park and nation-wide monitoring are similar and increasing (Auniņš et al. 2014).

The data collected was not sufficient for an objective assessment of habitat choice with regards to bush removal or the combination of all management activities. With regards to pastures, the choice is not significantly different from random, which might imply that pastures are just as important as any other habitats in the project area. Since most of the pastures were established before the project, it was not possible to assess the situation before, but since management took place in the most overgrown areas, the Corncrake might have benefited from the change from unsuitable habitat to suitable and this could be related to the population increase (Table 6). At the same time, it must be taken into account, that pastures are not homogeneous within the habitat, therefore can have differing suitability for the Corncrake (Abaja 2013). In previous species research in Latvia, it has been found that pastures are not of the most preferred habitat for the Corncrake with regards to singing male densities and interpreted hierarchical suitability (Keišs 2006b). It must be taken into account that habitats in the nature park are managed for biodiversity, not for financial gain, therefore habitats can be more suitable for species' breeding, and in the project area the density of singing males detected is two to three times higher than in the research compared.

In those years where the Corncrake density was greater in the project territory than in the nature park as a whole, one can see (Table 11) a greater density in the managed habitats. This could indicate that the landscape change (management activities) will have a positive influence on the Corncrake population in the future. Since management in the project is related to pastures, habitats become of the most importance to the Corncrake in the late breeding season (time of repeated breeding) when the surrounding grasslands are being cut (possibly many tens of kilometres away) but the vegetation in pastures is still suitable for breeding. Unfortunately the censuses were not designed for data collection during repeated breeding, therefore this hypothesis cannot be adequately tested.

At the same time, population trends for another specially protected bird species – the Great Snipe – are increasing rapidly and significantly. Since the trend in the nature park is declining, the population change could be related to management during the project.

Significantly declining population trends are found for species related to bushes – Blyth`s Reed Warbler and Common Grasshopper Warbler – and Spotted Crake. Changes in both Warbler species can be related to management's influence on nocturnal farmland bird species. Unfortunately those species were not counted in control routes, therefore it is impossible to estimate population changes in the nature park as a whole. The trends for the Spotted Crake in both control and project census routes are visually similar. The change in the Spotted Crake trend in the nature park is not significant, probably because of a short-term population increase before the beginning of the project. It is assumed, that those populations are being affected by factors operating outside the project territory and has a similar effect throughout the nature park.

References

Abaja R. 2013. Sugas biotopu modelis griezei (Crex crex L.) Dvietes palienes dabas parkā. Atskaite.

Auniņš A., Keišs O., Reihmanis J., Avotiņš jun. A. 2014. Fona monitorings: Putni. Gala atskaite par 2014. gadu.

Keišs O. 2006a. Naktsputnu uzskaišu metodika: http://www.lob.lv/download/Naktsputni_lauksaimnieciba_met.doc

Keišs O. 2006b. Lauksaimniecības pārmaiņu ietekme uz griezes *Crex crex* (L.) populāciju Latvijā: skaita dinamika, biotopu izvēle un populācijas struktūra. Doktora disertācija. Latvijas Universitātes bioloģijas fakultāte. 87 lpp.

Manly B. F. J., McDonald L. L., Thomas T. L., McDonald T. L., Erickson W. P. 2002. Resource Selection by Animals. Statistical Design and Analysis for Field Studies. Second Edition. Kluwer Academic Publishers, Dordrecht, 221 pp.

van der Meij T. 2007. BirdSTATs. Species Trends Analysis Tool (STAT) for European bird data. Manual. Bioland Informazie, Oegstgeest/Niederlande. 28 pp.

Račinska I., Klepers A. 2008. Latvijas palieņu pļavu atjaunošana Eiropas Savienības prioritāro sugu un biotopu saglabāšanai. 2004–2008. Projekta pārskats. Latvijas Dabas fonds.

Appendix 1

The number of Corncrakes Crex crex counted in the Dviete floodplain nature park 2011–2015

Route[1]	2011					2012					2013				
	A	B	C	D	E	A	B	C	D	E	A	B	C	D	E
K-1	13	12	13	12,5	3,88	5	9	9	7,0	2,17	11	10	11	10,5	3,26
K-2	4	9	9	6,5	2,01	6	6	6	6,0	1,85	14	9	14	11,5	3,54
K-3	6	9	9	7,5	1,82	14	17	17	15,5	3,77	15	10	15	12,5	3,04
K-4	26	37	37	31,5	3,60	51	44	51	47,5	5,43	50	40	50	45,0	5,14
K-5	13	10	13	11,5	4,44	14	2	14	8,0	3,08	7	6	6	6,5	2,50
P-1	2	14	14	8,0	1,57	3	5	5	4,0	0,79	14	14	14	14,0	2,75
P-2	3	5	5	4,0	0,78	4	6	6	5,0	0,97	5	12	12	8,5	1,66

Route[1]	2014					2015				
	A	B	C	D	E	A	B	C	D	E
K-1	14	8	14	11	3,42	11	17	17	14	4,35
K-2	12	11	12	11.5	3,55	12	16	16	14	4,32
K-3	18	11	18	14.5	3,53	10	15	15	12,5	3,04
K-4	33	37	37	35	4,00	26	36	36	31	3,54
K-5	8	5	8	6.5	2,50	14	12	14	13	5,00
P-1	9	13	13	11	2,16	15	11	15	13	2,55
P-2	1	3	3	2	0,39	9	14	14	11,5	2,25

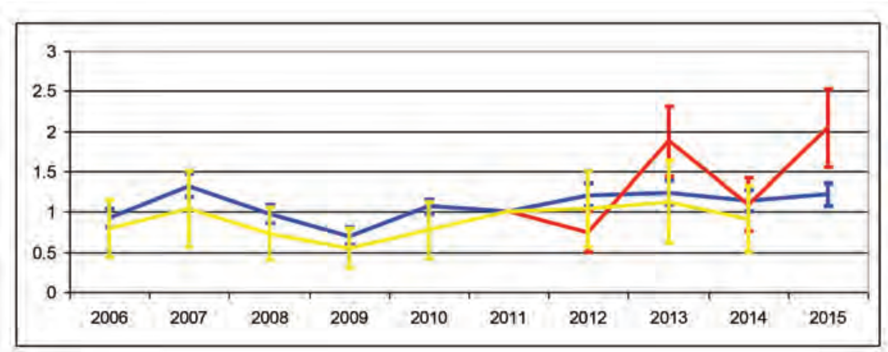
[1] Routes as in Fig. 1
A – first census
B – second census
C - max. Corncrake number per census
D - average Corncrake number per census
E - mean Corncrake number per census kilometer

Appendix 2

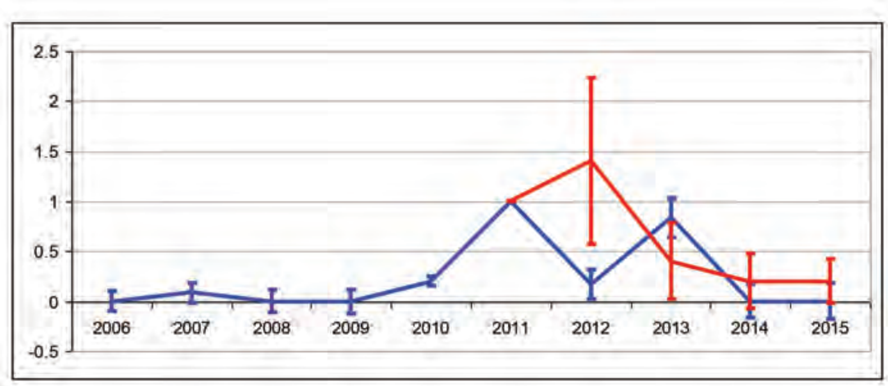
Nocturnal bird population trends in Dviete floodplain nature park 2006–2015

Legend:

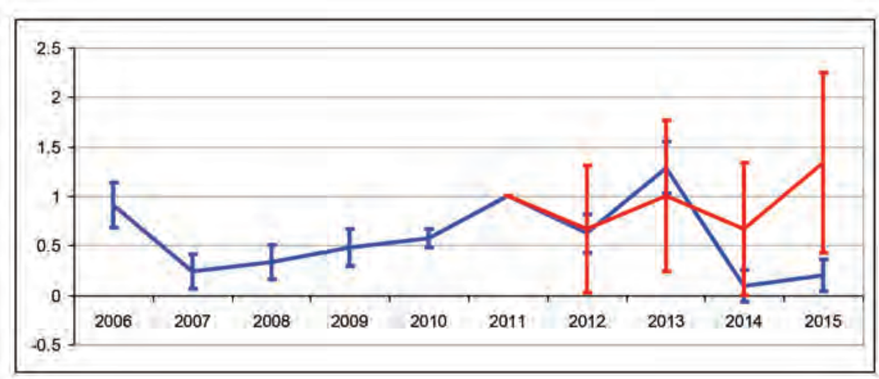
- the blue line shows the population index in 'control' routes, the red line – the population index in the project area, while national Corncrake monitoring index values are marked with the yellow line;
- error bars of the respective colour show the standard error for the population index



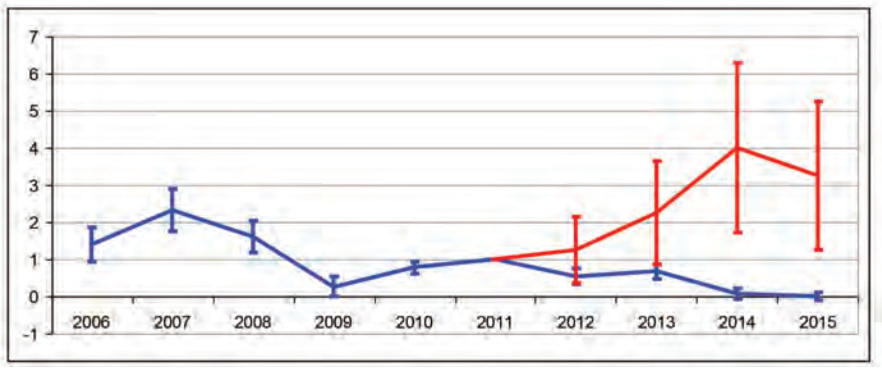
Corncrake Crex crex. The trend in the entire nature park (since 2006) – moderate increase ($p<0.01$), in the project area (since 2011) – steep increase ($p<0.05$), national population is stable.



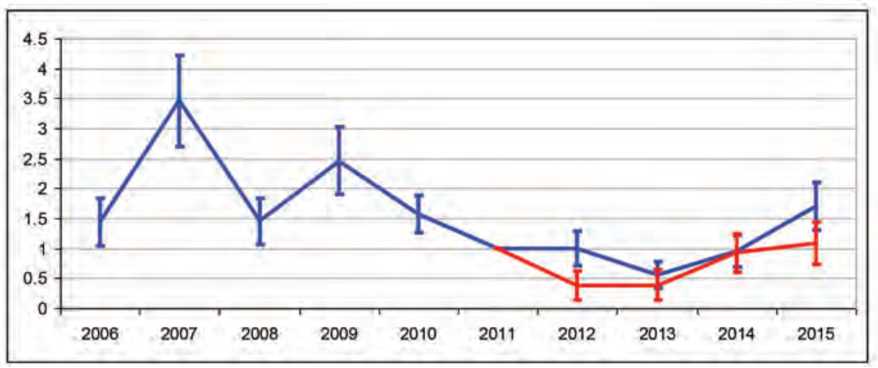
Spotted Crake Porzana porzana. The trend in the entire nature park (since 2006) – uncertain, in the project area (since 2011) – steep decline ($p<0.05$).



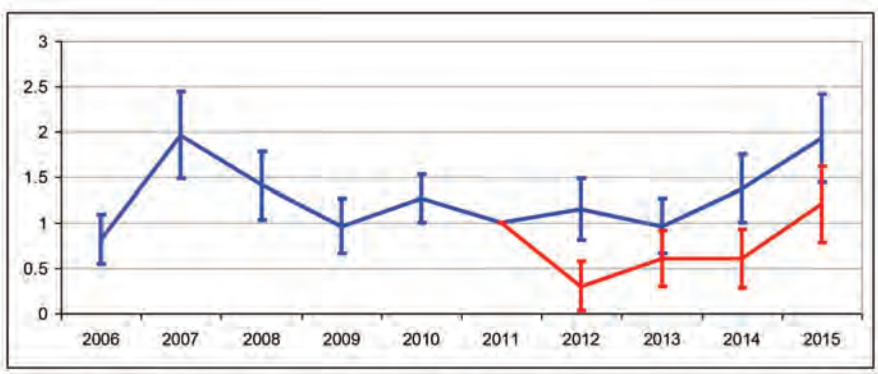
Common Snipe Gallinago gallinago. The trend in the entire nature park (since 2006) – uncertain, in the project area (since 2011) – uncertain.



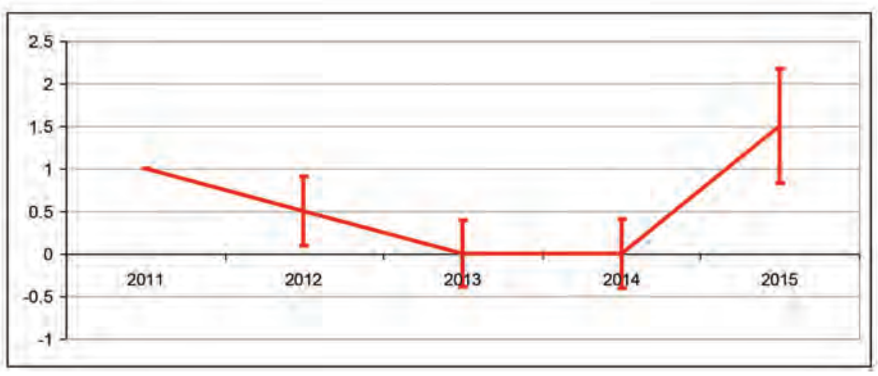
Great Snipe Gallinago media. The trend in the entire nature park (since 2006) – steep decline (p‹0.01), in the project area (since 2011) – steep increase (p‹0.05).



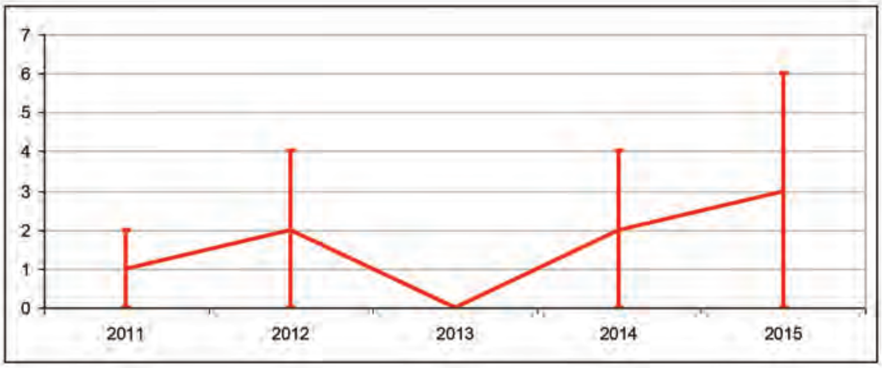
Common Grasshopper-Warbler Locustella naevia. The trend in the entire nature park (since 2006) – steep decline (p‹0.05), in the project area (since 2011) – uncertain.



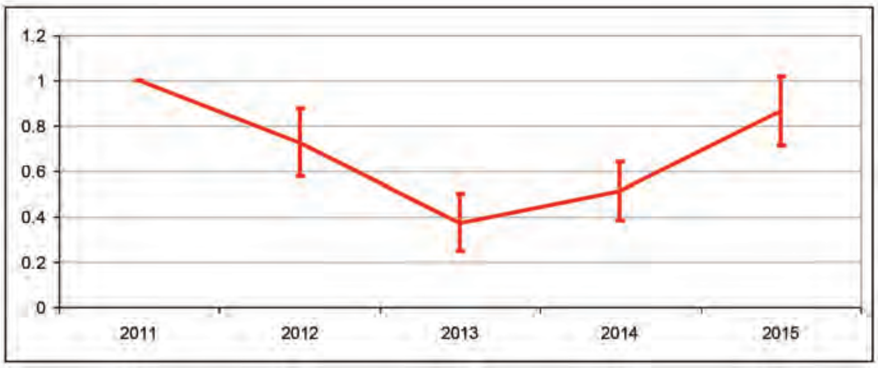
Eurasian River Warbler Locustella fluviatilis. The trend in the entire nature park (since 2006) – uncertain, in the project area (since 2011) – uncertain.



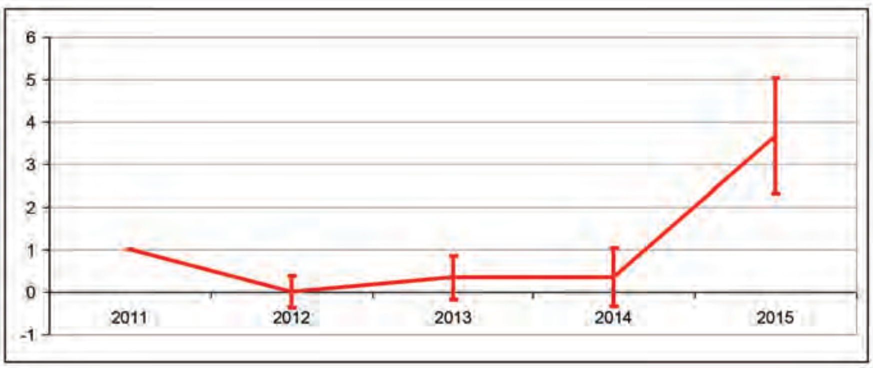
Common Quail Coturnix coturnix. The trend in the project area (since 2011) – uncertain.



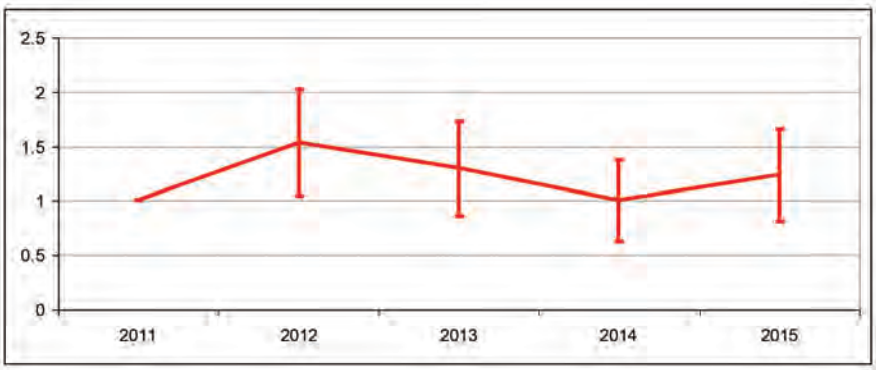
Eurasian Woodcock Scolopax rusticola. The trend in the project area (since 2011) – uncertain.



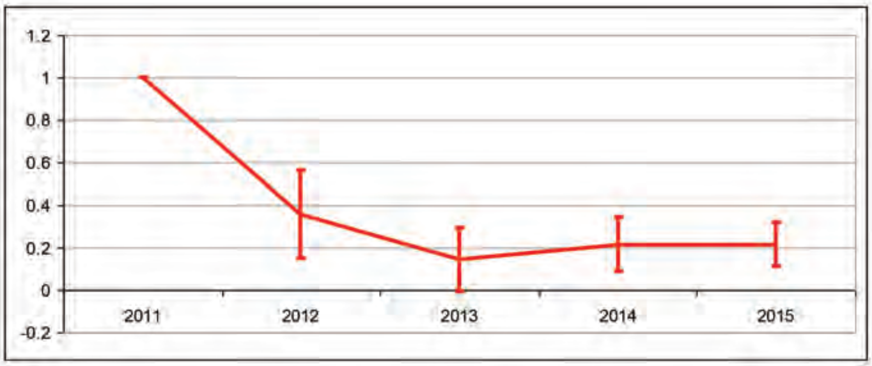
Thrush Nightingale Luscinia luscinia. The trend in the project area (since 2011) - uncertain.



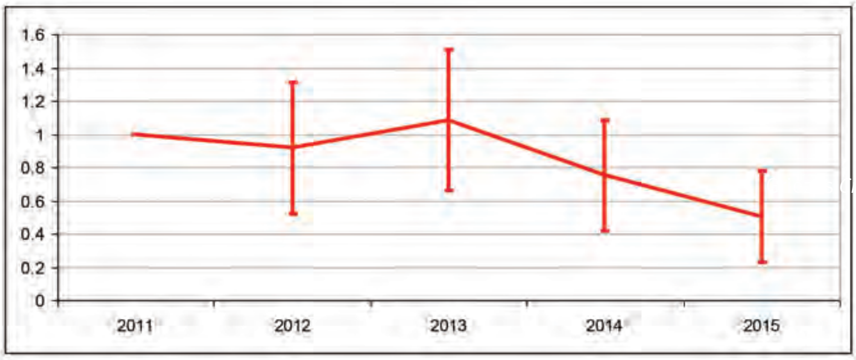
Savi's Warbler Locustella luscinioides. The trend in the project area (since 2011) – uncertain.



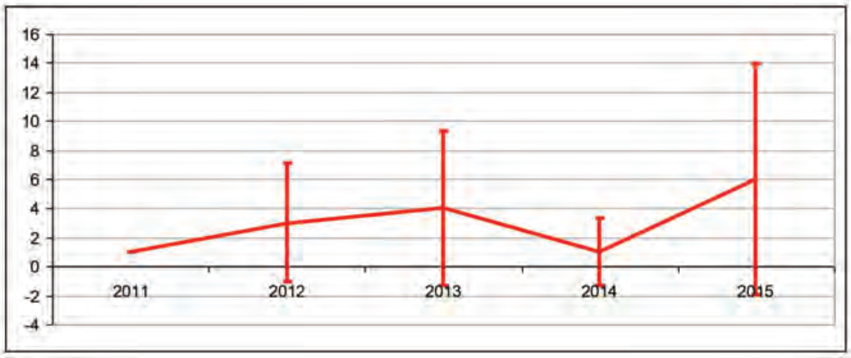
Sedge Warbler Acrocephalus schoenobaenus. The trend in the project area (since 2011) – uncertain.



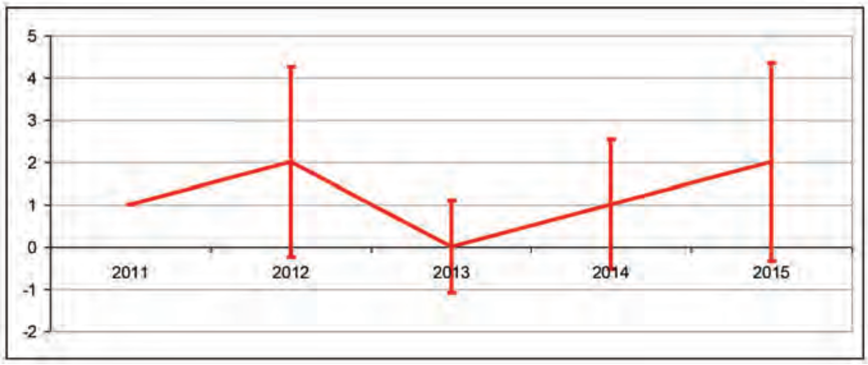
Blyth's Reed-Warbler Acrocephalus dumetorum. The trend in the project area (since 2011) – steep decline (p‹0.01).



Marsh Warbler Acrocephalus palustris. The trend in the project area (since 2011) – uncertain.



Common Reed-Warbler Acrocephalus scirpaceus. The trend in the project area (since 2011) – uncertain.



Great Reed-Warbler Acrocephalus arundinaceus. The trend in the project area (since 2011) – uncertain.

Corncrake *Crex crex* suitable habitat modelling in the Dviete floodplain Nature Park

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Summary

Landscape modelling is becoming increasingly important for long-term decision making in nature conservation efforts and environmental research. Both in Latvia and the European Union, the Corncrake *Crex crex* is an especially endangered bird species whose conservation requires planned management of the species protection in the scale of landscape. The high resolution data acquired as part of the LIFE+ project 'Restoration of Corncrake habitats in Dviete floodplain Natura 2000 site' to develop and compare DSS (decision support system) and ENFA (ecological niche factor analysis) models. Out of the two developed habitat suitability maps, the ENFA model was better at representing in situ conditions. The use of its methodology for future habitat suitability status evaluation and action planning for Corncrake conservation is recommended. To use DSS modelling methodology, further research about Corncrake requirements and improvements in the model's ability to represent in situ conditions are necessary.

Key words: suitable habitat model, Decision Support System (DSS), Ecological Niche Factor Analysis (ENFA), remote sensing, Corncrake *Crex crex*

Introduction

The Corncrake *Crex crex* is a protected bird species in Latvia and the European Union, whose original floodplain habitats have been significantly altered (Atsma 2006). As a result of human induced land-use change, floodplains have become an increasingly rare habitat (Auniņš 2010). Consequently, the increasing intensity of agriculture or the overgrowth of bushes and afforestation of abandoned lands endangers Corncrakes through the destruction of their natural habitats (Keišs 2006). Therefore, suitable habitat modelling is crucial for the conservation of the species, particularly in especially protected nature territories, such as the Dviete floodplain Nature Park.

Landscape modelling information is an important source for sustainable long-term decision-making (Osborne et al. 2001; Sanderson et al. 2002; Suchant et al. 2003; Jooss et al. 2009; Gaucherel et al. 2010). Suitable habitat modelling is necessary to identify potentially unknown Corncrake breeding areas. Modelling helps to determine the scale and direction of the required human management to restore suitable habitats for the endangered bird. Compared to traditional Corncrake counts, the use of remote sensing for habitat modelling offers a faster, more comprehensive and realistic evaluation of the species' habitat state. The spatial characteristics of the landscape that determine habitat suitability in the model can be used to assess Corncrake status not only at the local population level, but also on a wider scale. Several approaches for landscape modelling exist. One of the most widely used methods is modelling according to the Decision Support System (DSS) principles. A DSS model is created based on the accumulated information about the species' habitat requirements with expert knowledge used to supplement the missing information (Opdam et al. 2002). This modelling approach is suitable for species, whose habitat requirements are well studied (Store, Kangas 2011).

Several alternatives exist for the modelling of less studied species, however the majority of them require precise data about the species' presence and absence within the studied area. Ecological Niche Factor Analysis (ENFA) methodology is appropriate for cases where information about the species' presence is known only for separate locations. Corncrake counting data corresponds to the described case. To model the habitats, *Biomapper*, a software created to enable ENFA for suitable habitat identification using remote sensing methods (Strubbe, Matthysen 2008), was used.

The research objective was to develop a Corncrake habitat suitability model, which was used to determine habitat suitability status in the Dviete floodplain

Nature Park. To determine the precision of each model, suitable habitat models were created using both DSS and ENFA methods.

The concrete actions for the accomplishment of the project's objective:
1. Developing and testing DSS and ENFA models for Corncrake habitat suitability evaluation;
2. Comparison of both models;
3. Development of a practically usable habitat suitability map for the Dviete floodplain Nature Park's Corncrakes using the more precise and reliable model.

Methods

Project area

The Corncrake habitat suitability model was created as part of the European Union's LIFE+ project 'Restoration of Corncrake habitats in Dviete floodplain Natura 2000 site' (No. LIFE 09 NAT/LV/00237) in the Dviete floodplain Nature Park.

The Dviete floodplain Nature Park is located on the left bank of the Daugava River in southeast Latvia (Figure 1). According to the administrative territorial breakdown, the Nature Park is located in Ilūkste's district's Bebrene's, Dviete's and Pilskalne's parishes. Part of the park is also located in Jēkabpils' district's Rubene parish.

The Nature Park is mainly situated in the Dviete River valley and its floodplain, and the downstream floodplain of Ilūkste River. The total park area is 4989 hectares. The Dviete floodplain Nature Park was created in 2004, concurrently acting as a Natura 2000 site aiming to protect and improve the territory's floodplain grasslands and the biodiversity of the breeding and migrating birds. The Dviete floodplain and its surrounding area is one of the most important Corncrake breeding sites in Latvia.

Data collection and analysis

Remote sensing data

The Institute for Environmental Solutions' airplane equipped with a LIDAR laser scanner and CASI1500 visible and near infrared light sensors flew over the project territory to collect remote sensing data. Aviation based remote sensing data provides especially high resolution spatial data (Wulder et al. 2004). The acquired and used raster map pixel size in this study corresponds to 20 to 50 centimetres in nature.

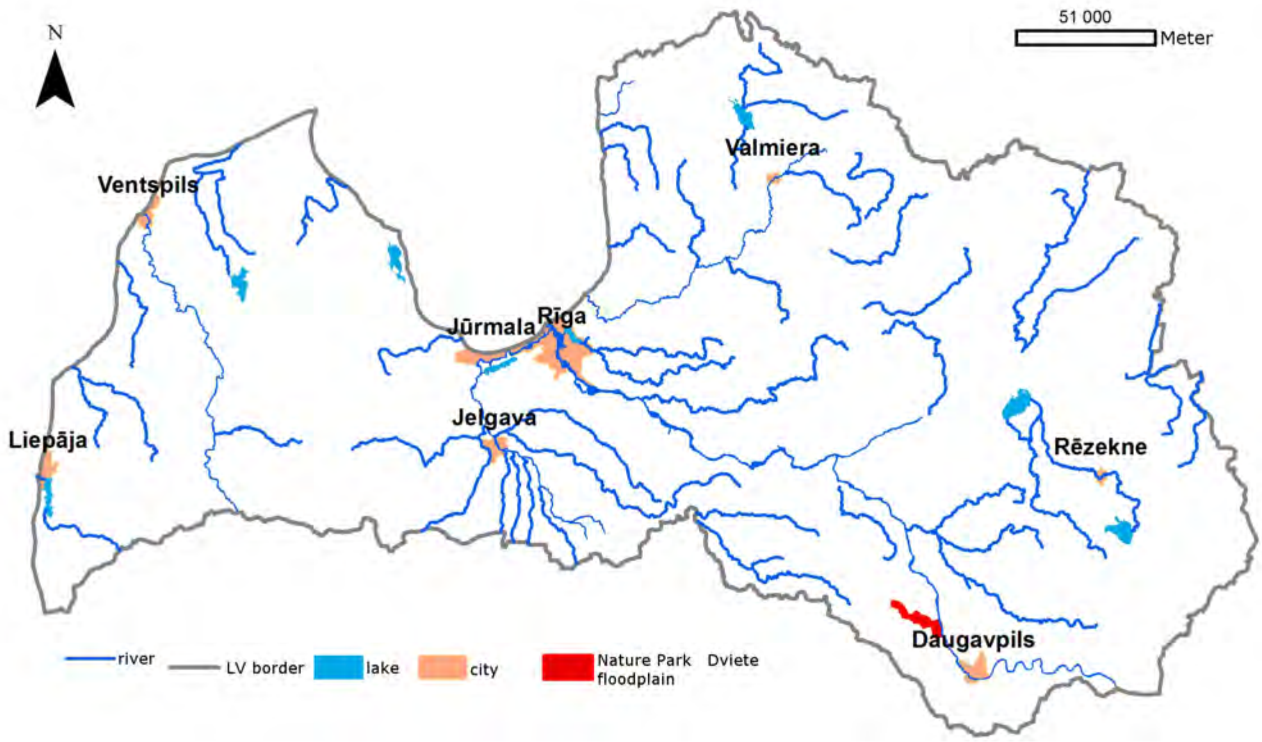


Figure 1. *The location of Nature Park 'Dviete floodplain' in Latvia.*

The LIDAR laser scanner data was used to develop a digital elevation model (DEM) and a normalised digital surface model (NDSM). As part of the Corncrake suitable habitat modelling, DEM was used to develop a floodplain distribution map. Floodplain grasslands are the original habitats for Corncrakes prior to human induced land-use change for intensive agriculture (Atsma 2006). The average low water period water level in Dviete River is 85 metres above sea level, however the average maximum flood water level is 90 metres above sea level (Račinskis 2005). According to the relief elevation information, the DEM model map was reclassified as a simplified floodplain map. Floodplains in this map were territories that were between 85-90.4 metres above sea level and were adjacent to the Dviete or Ilūkste rivers. The floodplain map was used to create the DSS model. The data was prepared and processed in ArcGIS software.

NDSM data was used to create forest and shrub distribution maps. NDSM data contains information regarding the elevation data of all objects above the ground surface level. The model's data is useful for vegetation and building height assessment. A forest distribution map was prepared using ArcGIS to reclassify NDSM and reflects data about objects whose elevation is 8–50 metres above ground level. A forest distribution map was necessary to create the DSS model and exclude forests as entirely unsuitable habitats for Corncrakes.

Shrub distribution map was also developed in ArcGIS using NDSM data. Shrubs were defined as objects that are 2-8 metres above ground level. Without additional data, it is impossible to separate trees and shrubs from buildings in NDSM data. To distinguish and exclude buildings from the data between the similarly high shrubs, Rural Support Service (RSS) data about the exact location of buildings in the project area was used.

Shrubs have a dual impact on Corncrakes. Isolated shrubs attract Corncrakes when they are choosing their breeding habitats (Wettstein et al. 2001; Schäffer, Koffijberg 2004). However, wide distribution of shrubs that arises during meadow overgrowth is not appropriate for Corncrake requirements (Keišs 2005, 2006). Therefore, sparsely and densely growing shrubs were separated in the DSS model. Sparsely growing shrubs were defined as those occupying an area of less than 0.1 hectares. Densely growing shrubs were defined as those occupying an area of more than 0.1 hectares. Consequently, sparsely growing shrubs were applied a positive distance value, that was progressively decreasing up until 10 metres interval from the shrub. Meanwhile, the densely growing shrub map attributed a negative impact zone of 10 metres. These methods were used to develop sparsely and densely growing shrub maps for the DSS model creation.

Hyperspectral data was gathered using 15 channels (total spectral coverage: 442.8 ± 6.0 nanometres to 885.5 ± 4.8 nanometres) of the CASI 1500 sensor. Hyperspectral data depending on their interrelated spectral channel combination provide comprehensive information about the vegetation status, vegetation type, growing conditions, ground use and other conditions (Ben-Dor et al. 2002; Lu, Weng 2007; Thenkabail et al. 2012). The correlation analysis between 15 channels was performed to model Corncrake habitat suitability. Following correlation analysis, the three most different channels that potentially contain the most information and are displayable in red, green and blue light channels according to human colour perception were chosen. This task was performed by the Institute of Electronics and Computer Science specialists. The acquired three channel raster images were used for ENFA model creation.

Land use data

O. Keišs (2005, 2006) research revealed that land management of agricultural lands crucially impacts habitat suitability for the Corncrake. Polygon spatial data about land-use in agriculture and grassland of high biodiversity value distribution was acquired from the RSS. This data was necessary for the development of DSS model. The considerably more detailed information about land-usage necessary for ENFA model was provided by hyperspectral data.

Data about agricultural land type prior to their inclusion in DSS model were categorised in suitability classes. Class distinctions were based on habitat suitability classes developed by the Corncrake expert O. Keišs according to his PhD thesis (2006). RSS culture classification is more detailed, therefore RSS class types were equated with O. Keišs' classification (Table 1). Using ArcGIS, the five suitability classes were calculated using class values proportionally to O. Keišs' classification to reclassify spatial data of agriculture land types accordingly.

In territories that are registered as grasslands of high biodiversity value, one of the requirements in the prior European Union planning period was to ensure land management with extensive grazing or late mowing (Cabinet Regulation No 295 of 23 March 2010). Late mowing is advised as one of the most appropriate grassland management methods for territories were Corncrakes are present (Green 1996; Atsma 2006; Gustafson 2007). Therefore, for the development of DSS model it was assumed that biologically valuable grassland territories are comparatively more suitable for Corncrakes.

The most appropriate grassland management method is mowing, because usually in grazing the developed vegetation is too short for Corncrake nesting and

Table 1.
Agricultural land classification types included in the Decision Support System model according to habitat suitability for the Corncrake, created using classification and values of O. Keišs's PhD thesis (2006) and adjusted to the model according to Rural Support Service (RSS) data classification.

	Habitat suitability classes	Model class suitability value	Land-use types in O. Keišs' PhD thesis	Corncrake count per area in O. Keišs' PhD thesis	RSS agricultural land type
1.	Suitable	0,43988	Set-aside grassland; Uncultivated grassland; Set-aside arable land	3,05 – 2,73	620; 710; grasslands not registered in RSS data
2.	Slightly less suited	0,24429	Cultivated grassland; Other habitat	1,61 – 1,60	720; 730; 930; 989
3.	Less suited	0,18950	Winter cropland	1,35 – 1,14	112; 121; 132; 150; 160; 610
4.	Rarely suited	0,10807	Spring cropland; Orchard; Berry bushes	0,72 – 0,70	111; 131; 140; 212; 410; 420; 530; 730; 910; 911; 921
5.	Very rarely suited	0,01826	Vegetable land	0,12	211; 810; 820; 840; 841; 843; 846; 847; 926

habitation (Atsma 2006, Gustafson 2007). Therefore, for DSS model data regarding grazing intensity within the project territory was searched. The only available spatial data was polygon spatial data delivered by E. Račinskis regarding the grazing practices implemented before and within this LIFE+ project to maintain grasslands. To improve the functionality of the model elsewhere, the DSS model incorporated this incomplete grazing distribution data.

Presuming that Corncrakes avoid habitats in close proximity to roads, data regarding the road network in the project territory was necessary for the development of a habitat suitability model. Following consultations with Corncrake experts and counters Oskars Keišs, Jānis Reihmanis and Edmunds Račinskis it was determined that the impact of roads varies according to the intensity of traffic. Therefore, roads were classified in three categories:

- First category roads – paved roads, that are characterised by the most intensive traffic flow (Midpoint – 30);
- Second category roads – gravelled roads, whose width is 5–7 meters and are characterised by moderate traffic flow (Midpoint – 20);
- Third category roads – gravelled roads and roads without cover that are narrower than 5 metres and have the lowest impacting traffic intensity (Midpoint – 10).

For every road category, a separate spatial data layer was created, which was assigned proximity measures from road axis. In the developed raster map, pixels closer to the road had a lower value than pixels farther from the road. In the ENFA model the different road category value changes were not limited by assigning a specific distance at which the proximity of the road has a negative impact. However, for DSS model each separate road category was assigned a different negative impact distance. It was achieved by using ArcGIS tool Spatial Data Modeller Tools/ Fuzzy Logic/Fuzzy Membership/Large assigning different Midpoint values (indicated in road category classification previously). The highest value indicates an observation of a negative impact farther away from the road. This led to the development of three different road impact maps, where pixel values were summed and afterwards led to the creation of a single road impact map.

Corncrakes are rarely seen in proximity to human populated spaces, however Corncrake males have been observed utilising the walls of nearby buildings as amplifiers for their vocalisations (Schäffer, Koffijberg 2004). Farmsteads are the most common residential types within Dviete floodplain Nature Park territory. The presence of humans and domestic animals corresponds to increased breeding risks for the Corncrakes, therefore it was necessary to develop a map reflecting this habitat suitability impacting factor. Similarly as with roads, data of building location was used to create a raster map, where pixel values corresponded to proximity from buildings. Pixels farther from buildings were classified as more suitable than those closer. The map showing the negative impacts of buildings which was developed using ArcGIS was used for DSS and ENFA modelling.

According to consultations with Corncrake experts, landscape elements connected with water – lakes, rivers, ponds and ditches have a neutral impact on Corncrake habitat choice, therefore these landscape elements were not part of the proximity impact map. However, preparation of a water habitat map was necessary, to enable the exclusion of these habitats as completely unsuitable for the Corncrake. Raster map that contains information about the territory's water habitats was prepared using ArcGIS.

Corncrake counting data preparation and processing

As part of the project, Corncrake counting was performed by walking through a predetermined route and marking the areas in the map where Corncrake male vocalisation was heard. Two types of routes were selected. One was for Corncrake monitoring needs as part of the project requirements. These routes were situated in areas where project activities were planned. To increase the relevance of the data, Corncrake monitoring data of the routes implemented in the previous LIFE+ project 'Restoration of Latvian floodplains for EU priority species and habitats' were selected for usage in the modelling. To incorporate data about the whole Dviete Floodplain Nature Park territory, in addition to airborne remote sensing data collected in 2011, ground Corncrake counting was performed also in 4 routes that cover the largest agricultural areas where prior Corncrake monitoring had not been performed. Data from 13 Corncrake counting routes was used for modelling. Information regarding detailed Corncrake counting routes and monitoring methods can be observed in Latvian Ornithological Society (Ķerus et al. 2014) and Institute for Environmental Solutions (Reihmanis 2011) monitoring reports.

Corncrake vocalisation is usually heard no further than 1.5 kilometres (Schäffer, Koffijberg 2004). However, according to Corncrake monitoring experts E. Račinskis and J. Reihmanis precise mapping of Corncrake can be made in locations within a 100 metre proximity from the monitoring route, while locations further than 500 metres from the route are deemed to be too imprecise for suitable habitat modelling and therefore are excluded from data analyses. As a whole only three points were excluded (0.77 % from the Corncrake counting data).

To test whether all the registered Corncrake locations in a 500 metre distance away from route can be used for modelling, modelling was done twice. Firstly, with all data points in the 500 metre distance, while the second time only with information from the closest 100 metres. Afterwards, the two models were compared using an average pixel value of 50 metre radius around the observed locations. The average pixel value described habitat suitability, while the 50 metre radius described core area of Corncrake male vocalisation site which average radius is 48 metres according to previous other researches (Skliba, Fuchs 2004). A map of Corncrake locations in a 50 metre radius marked polygons was used for modelling.

After ENFA modelling method (see below) the 100 and 500 metre models were compared with the Mann-Whitney test in R software. It was concluded that there are no significant differences ($p = 0.6129$) between the models. Therefore, all stock points in the 500 metre range can be used for model creation.

All 13 Corncrake monitoring routes were surveyed twice. The first counting was done at the end of May and beginning of June (28.05.-11.06.) 2011, while the second time was a month later (18.06.-14.07.). Corncrakes usually have two broods in a season (Schäffer, Koffijberg 2004) and they have been known to change their breeding sites after the first nesting (Keišs 2006). The first count describes the sites of the first broods, while the second count – the sites of the second broods.

To ascertain whether two different habitat models are necessary for Corncrake habitat suitability assessment during the beginning and middle of the breeding season, separate models for the suitable Corncrake habitats were created using data gathered from the first and second count. The method of both model comparisons was similar to those of the 100 and 500 metre model comparisons. The results of the Mann-Whitney test indicated that the difference in Corncrake habitat requirements between the first and second count was not significant ($p = 0.2012$). Therefore, for model creation the unified data of the first and second count was used.

Corncrake counting data was used only for the ENFA model. Accounting for the aforementioned analysis conclusion, during the ENFA model creation both the first and second Corncrake vocalisation counting points within a 500 metre range from the monitoring route were used. A 50 metre radius polygon around each stock point was created in ArcGIS, assuming them to be Corncrake breeding sites with suitable habitat conditions. This data layer was converted in raster format, assigning a value of 1 for polygons and 0 for their surrounding area.

Model creation methods

The specificity of the data used prevented developing both models according to the same time period. The RSS database began registering polygon spatial data about agricultural land type in 2012, therefore the DSS model could only be created according to the situation in 2012. However, the most comprehensive Corncrake counting data for the ENFA model was from 2011. Additionally, the remote sensing data was also collected in 2011, therefore the developed ENFA model represents the situation in 2011.

The most time consuming part of model creation is data preparation and pre-processing according to species' requirements, chosen time period and data matching. Data of an equal spatial resolution (for the ENFA model – pixel size represents 5x5 metres in nature), coordination system (WGS 1984 UTM Zone 35N) and identical (pixel-in-pixel) image overlap were used for model creation. All of the following data outlined in the model descriptions was prepared according to this criteria. For data preparation and pre-processing ArcGIS was used. After data preparation in ArcGIS, its conversion from img to rst format and defining image (map layer) background values that are not part of model calculations were necessary for the creation of the ENFA model. This was accomplished in the IDRISI programme.

DSS modelis

The DSS model was created in ArcGIS, mainly using *Spatial Analyst* and *Spatial Data Modeller* tools.

For the creation of this model, seven raster data layers were used, each representing one of the Corncrake habitat suitability influencing factors (Table 2).

DSS modelling began with factor influence character, intensity and scattering property assignment, which was done with *Spatial Data Modeller Tools/ Fuzzy Logic/* ***Fuzzy Membership*** tools. If factors have a negative influence, then ***Large*** tool was used, if positive then ***Small***.

Each data layer prepared for the DSS model was attributed a degree of influence coefficient based on scientific literature and consultations with Corncrake experts (Table 2). Higher coefficient values represent a higher factor influence on habitat suitability as compared to other factors. The chosen coefficient values were assigned using *Spatial Analyst Tools/ Math/* ***Times*** tool.

After coefficient value assignment, all data layers were summed using the *Spatial Analyst/ Local/ Cell Statistics* (function – **SUM**) tool. Afterwards from the resulting data layer, all completely unsuitable habitat

Table 2.

Corncrake habitat suitability affecting factors. The importance of factors, type and parameters used in the Decision Support System model.

	Factors – their representation in map layers	**Degree of influence coefficient**	**Influence character**	**Spatial Data Modeller tool Fuzzy Membership Indicated factor intensities and scattering values**
1.	Distance from roads	0,40	Negative	Midpointa – 4; Spread – 5
2.	Distance from buildings	0,20	Negative	Midpoint – 5; Spread – 2
3.	Areas outside pastures	0,15	Positive	Midpoint – 1; Spread – 1
4.	Agricultural lands	0,10	Positive	Midpoint – 1; Spread – 1
5.	Distance from shrubs	0,07	Positive	Midpoint – 10; Spread – 1
6.	Grasslands of high biodiversity valuei	0,04	Positive	Midpoint – 1; Spread – 1
7.	Floodplain	0,04	Positive	Midpoint – 2; Spread – 1

elements – forests, dense shrubs, water bodies, roads and buildings were cut out. As a result, the subsequent data layer is the DSS model – habitat suitability map for Corncrakes, where higher pixel values represent more suitable conditions for Corncrakes.

DSS model testing was performed using visual analysis and comparison between the most suitable areas suggested in the map with the observed concentrations of Corncrakes during counting.

The ENFA model

The ENFA model was developed in the *BioMapper* programme. For the creation of this model the following data was used:

1. Species data – prepared and processed Corncrake counting data
2. Factor data (Note: hyperspectral data information used for the creation of this model was done by the Institute of Electronics and Computer Science):
    a. Primary mutually least correlating hyperspectral data spectral information;
    b. Secondary less correlating hyperspectral data spectral information;
    c. Less than the aforementioned, but considerably different hyperspectral data spectral information;
    d. Distance from first category roads;
    e. Distance from second category roads;
    f. Distance from third category roads;
    g. Distance from buildings;
    h. Distance from sparsely growing shrubs.

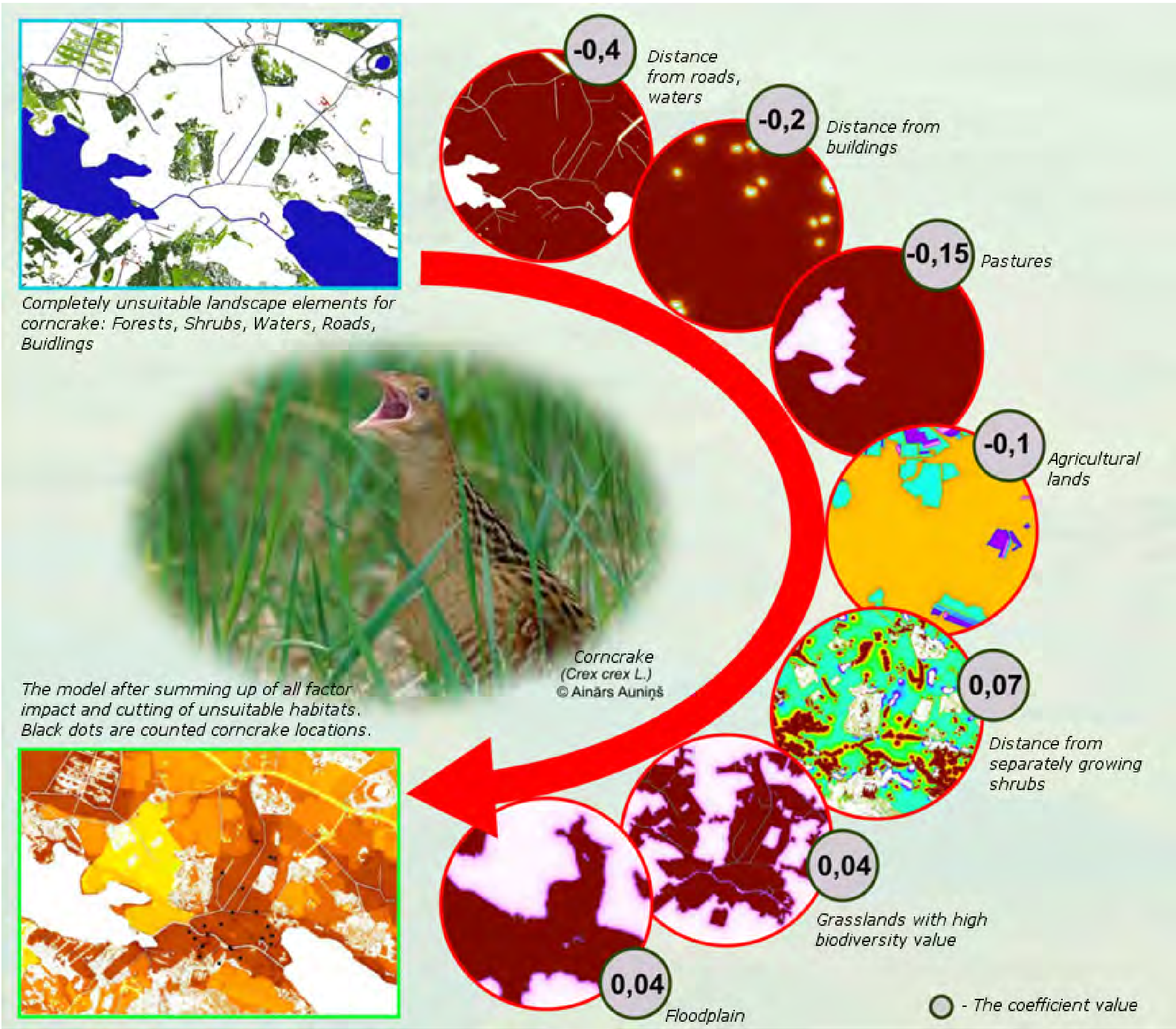


Figure 2. *The process of DSS modelling, beginning with the exclusion of completely unsuitable habitats, followed by factor map layer preparation and the assignment of a coefficient representing its influence, concluding with a visual comparison between the model suggestions and counting data of Corncrake distribution.*

Prior to modelling, factor data normalisation (Box-Cox transformation (Sokal, Rohlf 1995)) was performed, while also transferring data from integer to byte format, which reduced the amount of data and enables *BioMapper* use.

Afterwards all data (both factors and species) was tested to determine whether images correspond pixel-in-pixel, if they all have the same cells defined as background, if any of the factor maps does not contain only one or two value data. Only after testing does the programme allow modelling.

Firstly the covariation matrix and then ENFA was performed using the programme's automated modelling tool. ENFA is based on two attributes that determine the species' niche. Marginality indicates the extent of the species' requirements in choosing a suitable habitat (Hirzel et al. 2002; Ortega-Huerta, Peterson 2008) describing the real life conditions in which it is observable. Marginality values were selected analysing the overall value distribution in factor maps at those points where Corncrakes were registered. The second attribute is species' specialisation (Hirzel et al. 2002; Ortega-Huerta, Peterson 2008), whose values were acquired through the programme's analysis of distribution of found factor values at Corncrake vocalisation points. For example, if in these points, a profound domination of a factor combination is observed, then it can be concluded that the species has narrow specialised requirements. ENFA results provide marginality and specialisation values according to an important factor combination, which is used by the programme to create a habitat suitability map by appropriating these values to all other equivalent factor combinations corresponding to locations in factor map layers where Corncrake counts have not been made. ENFA is performed using the *Median* algorithm. This algorithm allows to find a median value of each factor for the Corncrake from the counting data by assuming that factor values at both ends diminish measuredly (Hirzel 2004).

After calculations, the programme offers to select the factors upon whose basis to create a habitat suitability map. Factor count is selected based on the broken-stick method of the regression curve showing the considerable changes in regards to the map's correspondence to ENFA results if one of the analysed factors is deselected (Fattorini 2005). The broken-stick method helps to create a habitat suitability map that is based on the species' most important factors. Following the selection of important factors, BioMapper offered to create a habitat suitability map for the Corncrake.

BioMapper's helper application *Data Manager* saved the acquired map in *jpg* (JPEG) format, which was then loaded in *ArcMap* for further model precision analysis. The acquired image had to be reset with a coordinate system to enable spatial pegging and conversion into raster format for map analysis.

For model testing the entire model process was repeated 10 times with the same ecological factor data, but with different Corncrake observation data points. The different data for model testing was acquired based on the coincidence principle, assigning new Corncrake vocalisation point numeration. Afterwards all points were sorted starting with the smallest number of Corncrake vocalisation points in the corresponding polygon with optimal, suboptimal or limited suitable habitat values, and distributing 10 equal point number classes. Consequently, each class accounted for 10 % of the overall counting data.

Correlation analysis in ArcGIS was performed for all 10 test model maps with the habitat suitability model map, which was created using counting data. The *Spatial Analyst Tools/ Multivariate/ Band Collection Statistics* tool was used for correlation analysis.

An additional comparison of 10 test models with the ENFA habitat model according to average values, standard deviation values, covariation and correlation indicators, which were acquired from the correlation analysis, was performed in ArcGIS. This correlation analysis was done according to the Pearson method in the R programme.

Suitable habitat map creation and the calculation of corresponding Corncrake suitability class occurrence density and count

The modelled map gradually reflects Corncrake habitat suitability from 0 to 100 % values. The following model use (Corncrake monitoring route planning and management planning for Corncrake conservation) is more useful in the form of a map, where all habitats are classified in four suitability categories – optimal, suboptimal, limited suitability and unsuitable habitats. Furthermore, it is advised to calculate the average Corncrake density on a single unit of area for each class type. Therefore, the Corncrake habitat suitability model was reclassified according to the four suitability classes in the following order:

1. Equalised the model's map values to ensure a smooth transition between pixel values by using ArcGIS' *Spatial Analyst Tools/ Neighborhood/* ***Filter*** tool with the low option. This was repeated two times.
2. After a repeated model value equalisation, the developed map was reclassified in three classes using *Spatial Analyst Tools/ Reclass/* ***Reclassify*** tool with the *Natural Breaks* option. During classification it is vital to assign little suited habitats with a value of 1, while the other habitats should be assigned values of 2 and 3.
3. To illustrate completely unsuitable habitats (forests, shrubs, buildings, roads, water bodies) in the reclassified map, the habitats value was

equated with zero. For this purpose DSS modelling data about completely unsuitable environments (defined value zero) was used by multiplying the reclassified map data with the unsuitable habitat data in ArcGIS using *Spatial Analyst Tools/ Math/ **Times*** tool.

To calculate the occurrence density of each suitability class and the average area occupied by a Corncrake male corresponding to each of the suitability classes, parts of the map that were more than 500 metres from the route were cut out. This was necessary to ensure that for the calculation, reliable territory data about Corncrake vocalisation can be used.

Afterwards for each habitat suitability class a *shp* polygon data layer was created using *Spatial Analyst's Conversion Tool/ From Raster/ Raster to Polygon* tool. For each newly created *shp* layer a new field in the attribute table was added, where using geometric calculation, the value of each field or polygon area was calculated.

To create a data base about the territorial dispersion of Corncrake vocalisation, a data layer with a 50 metre wide lane from Corncrake vocalisation points was developed. Class type which dominated in this lane was selected as the corresponding field attribute data. The actions were done in ArcGIS.

The total area and the number of Corncrakes of each field where at least one Corncrake vocalisation point was observed was entered into the data base. Calculations were done separately, by splitting the data of the first and second count. Afterwards, a corresponding time area for each class type vocalisation was calculated. Corncrake density on a single hectare was calculated by developing an appropriate proportion of field area with the number of Corncrakes detected in it. Finally, the average measures of the areas fields and standard deviations were calculated. The data base was created and calculations were performed in *Microsoft Excel*.

Results and discussion

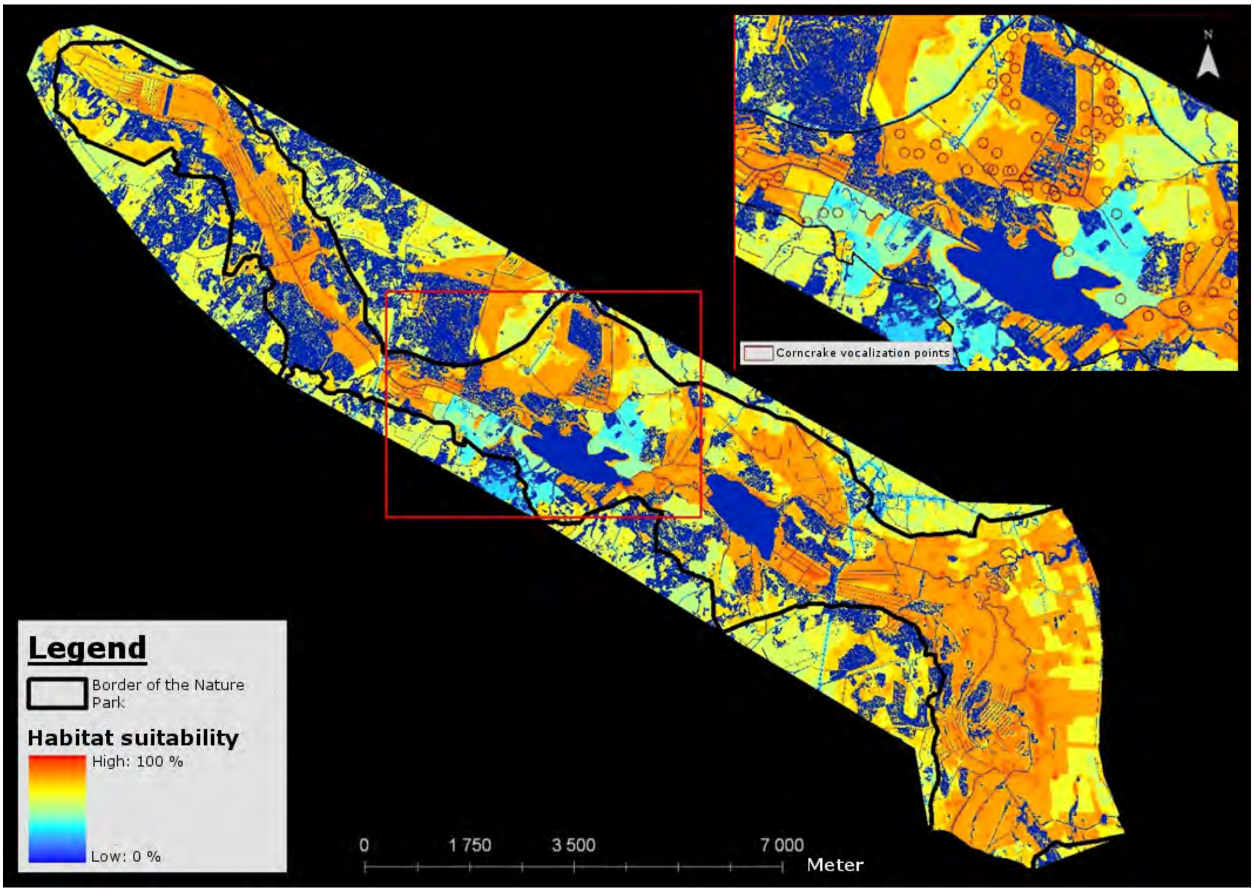
Modelling resulted in two Corncrake habitat suitability maps (Figure 3) – one based on the DSS model, the other on the ENFA model. As indicated in Figure 3, both models vary considerably in habitat suitability distribution and homogeneity. The observable differences are confirmed by a low correlation coefficient value of 0.13851. Mann-Whitney test results clearly indicate considerable differences ($p < 2.2 \cdot 10^{-16}$) between the two models.

Understanding which model is more reflective of the real life situation is seen in Figure 4. The DSS model appears to be overly optimistic, because all of the Corncrake vocalisation points occur in practically ideally suited habitats. However, in the ENFA model, habitat suitability is spread out over a wider range of suitability values. Here, the majority of data points are in habitats with 50-80% suitability, which is a more realistic reflection of the natural condition. Furthermore, the ENFA model habitat suitability value scale is more reflective of the normal distribution, which is typical for species ecological niche theory (Chase 2011). The DSS model scale is completely inadequate for the distribution of habitat suitability values.

One of the reasons why DSS turned out to be inadequate, could be the assumed factor influence values which were calculated according to those mentioned in previous Corncrake research on the strength of the impact of various factors. Exact factor importance could not be determined from previous research, therefore the selected values were based on the assumptions of Corncrake experts and model creators. The selected values appear to have been optimistic or the interacting relationship between influencing factors was modelled inaccurately due to a lack of appropriate research and understanding. The DSS model suitability is stressed for only well studied species (Angelstam et al. 2003; Uran, Janssen 2003).

The second reason that explains the lack of cohesiveness illustrated in Figure 3, could be the failure to include an important Corncrake habitat suitability affecting factor that is yet unknown. This would explain why the lack of its presence made the DSS model's results less related to the real situation, while the acquired hyperspectral data used for the ENFA model accounted for the factor's influence. A separate study analysing hyperspectral data together with Corncrake counting data and habitat conditions could reveal the identity of this as of yet unknown factor.

Several observable differences are displayed by the different models showing the same Skuķu Lake landscape (Figure 5). The light blue areas in the DSS model are locations where 'Konik' horses and cattle are grazing. The model had defined those territories with a diminished suitability value. However, in the ENFA model those areas do not stand out with a lower suitability index. Although grazing data was not included in the ENFA model due to a discrepant data format, the impact of grazing was indirectly included by the hyperspectral data showing the vegetation's spectral changes (Numata et al. 2007; Fava et al. 2009). From this information it can be concluded that the 0.4 cattle and horses per hectare grazing intensity in the Dviete floodplain Nature Park (animals remained in the enclosed grazing areas the whole year in 2011) did not have a significant negative influence on habitat suitability for Corncrakes. The lack of relevant data for the modelling method prevented an adequate analysis on the impact of intensive grazing. However, a number of studies have found a link between grazing



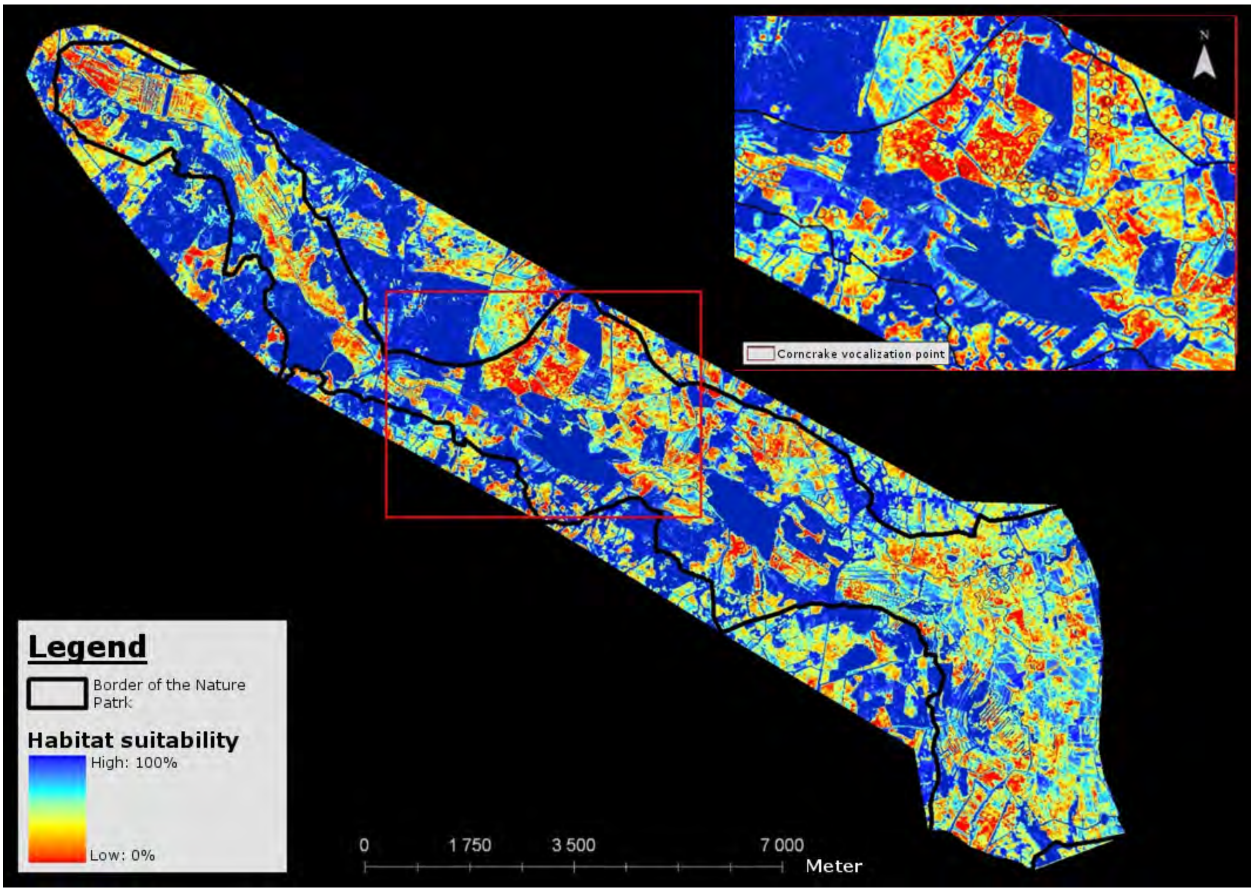


Figure 3. *The results of the DSS model (upper map) and the ENFA model (lower map) – Corncrake habitat suitability maps in the Dviete floodplain Nature Park and its surroundings. The highlighted area shows the visual model's comparison with Corncrake vocalisation points.*

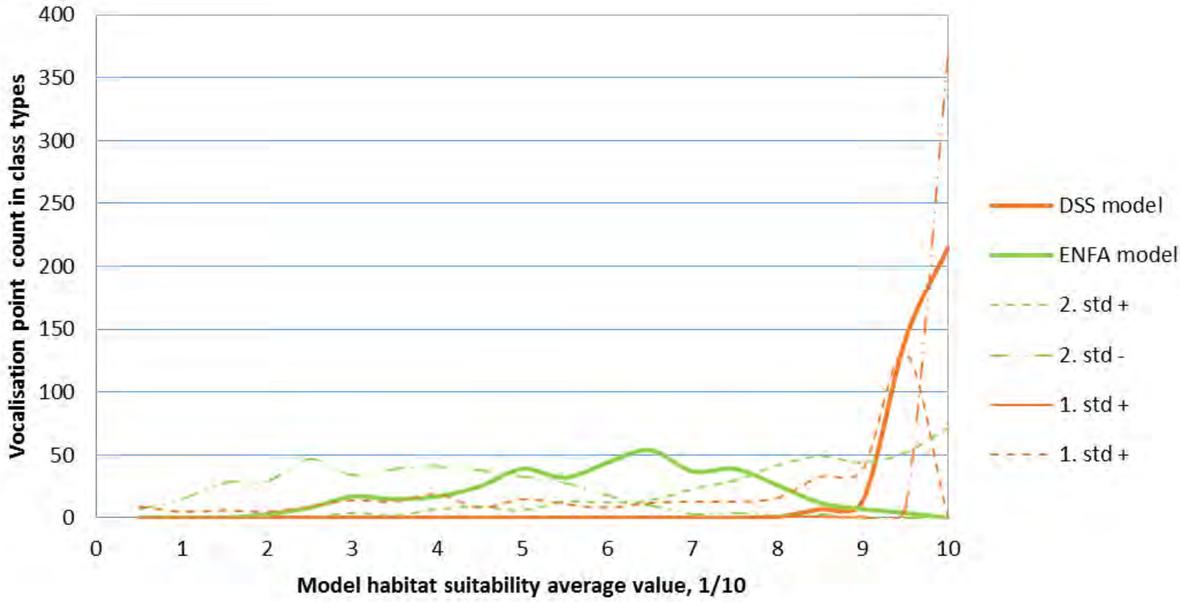


Figure 4. *Model habitat suitability distribution in a 50 metre radius around the vocalisation point. Higher suitability value indicates a more suitable habitat for the Corncrake. Legend: 1. Std+ (DSS model average value distribution with a positive standard deviation; 1. Std- (DSS model average value distribution with a negative standard deviation). Correspondingly the same for the ENFA model 2. Std+ and 2. Std-.*

and diminished habitat suitability for the Corncrake (Wettstein et al. 2001; Schäffer, Koffijberg 2004; Keišs 2006; van Weperen 2009).

The ENFA model shows a spatially wider negative impact of dense shrubs than the DSS model (Figure 5), thereby suggesting that grassland overgrowth with shrubs has a greater impact than previously thought. Consequently, an area of 30–50 metres is unsuitable for the Corncrake, not just the immediate proximity to dense shrubbery.

The influence of roads is reflected similarly in both models. Thus, the road categorisation system was an appropriate measure for Corncrake suitable habitat modelling. However, the negative influence of buildings and human habitats is larger in the ENFA model. This is best observed in the Dviete town area (Figure 6).

The Corncrake habitat suitability map

ENFA model results were deemed to be more representative of the actual situation. Its outcome was used to create a map with four habitat suitability classes, which would be used for informed decision-making in the Dviete floodplain Nature Park management and the planning of Corncrake conservation actions (Figure 7).

Average density of Corncrake occurrence and average area of habitat suitability per class is shown in Table 3. The percentage variations of habitat suitability allocations in Table 3 are based on BioMapper calculated class distribution proportionally to their dispersion in regards to the normal habitat suitability data. However, these figures only approximately describe Corncrake density and the corresponding area for each Corncrake male in the different habitat suitability classes. Most often the optimal class was in compact locations, while suboptimal classes were located in narrow lanes between optimal and limited suitability classes creating a large net-like shape. In rare cases, only some of the Corncrake vocalisation points with a 50 meter radius were completely included in suboptimal class fields. Most often the largest part of these point areas overlap on optimal habitat class fields. However, limited suitability class fields, similarly to optimal class fields, were featured by compact structure that is usually characteristic of dense shrubs, therefore optimal and limited suitability classes are more similar according to density and spatial coverage measures, although in fields of optimal habitats Corncrakes were observed three times more often than in fields of limited habitat suitability.

The required area necessary for one Corncrake male is similar to previous research results: 1.0–9.5 ha (Grabovsky 1993); 3–51 ha (Stowe, Dudson 1991). This proves the appropriateness of ENFA's modelling method for Corncrake habitat suitability modelling.

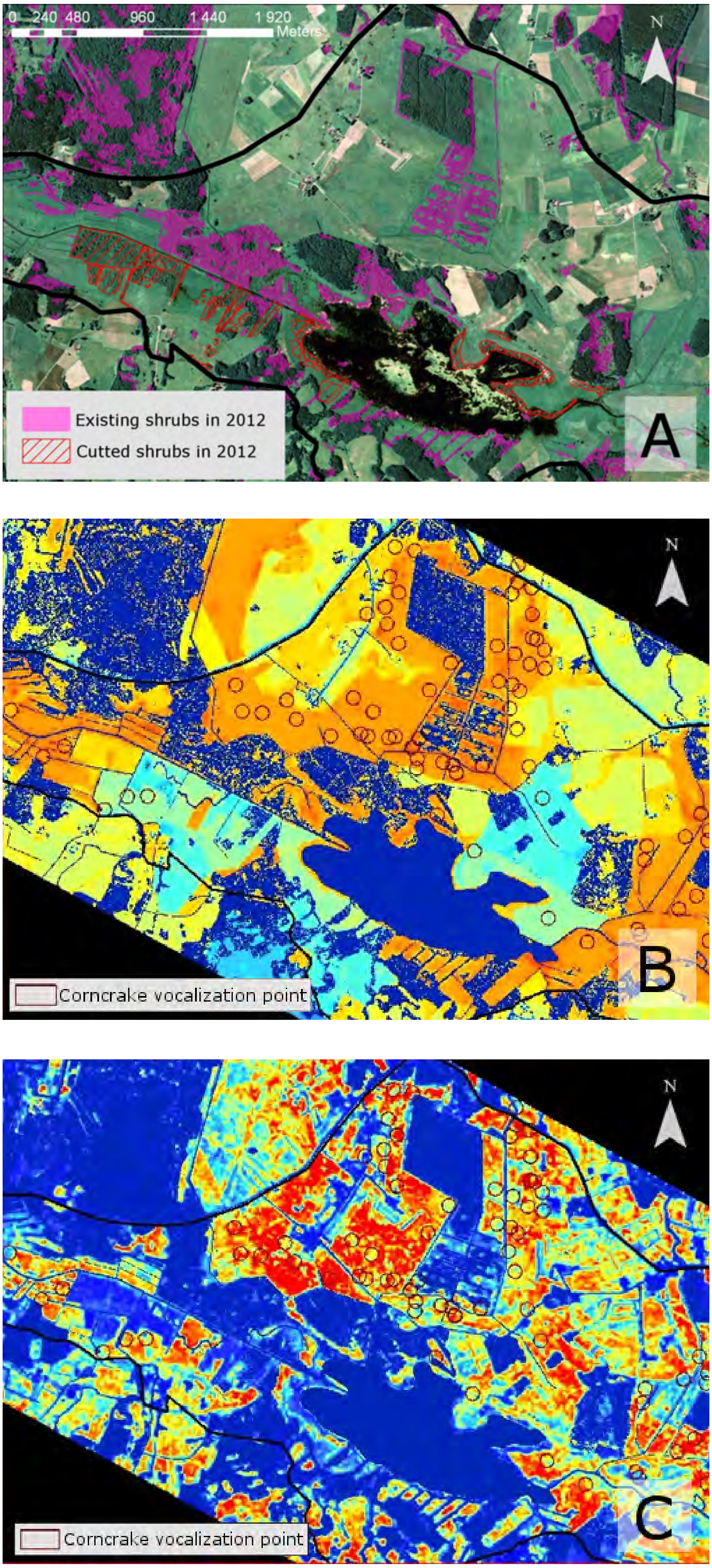


Figure 5. *Comparison of shrub impact between the models. A – orthophoto (pink filled spaces show existing shrubs in 2012, striped areas shows shrubs cut down in 2012); B – shrubs shown at the centre of a DSS model fragment (2012 data used); C – shrubs shown at the centre of ENFA model fragment (2011 data used).*

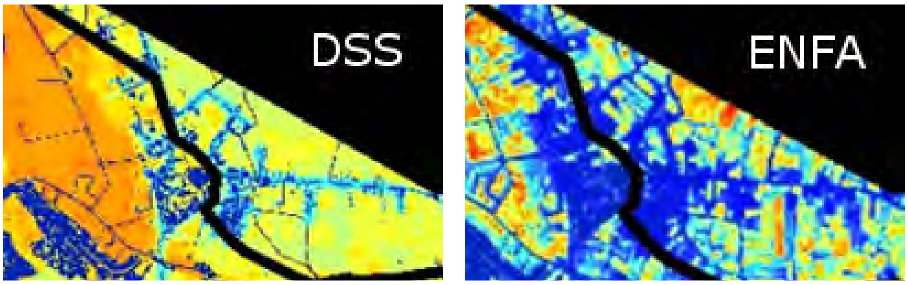


Figure 6. *Corncrake habitat suitability in the Dviete town area in DSS and ENFA model maps.*

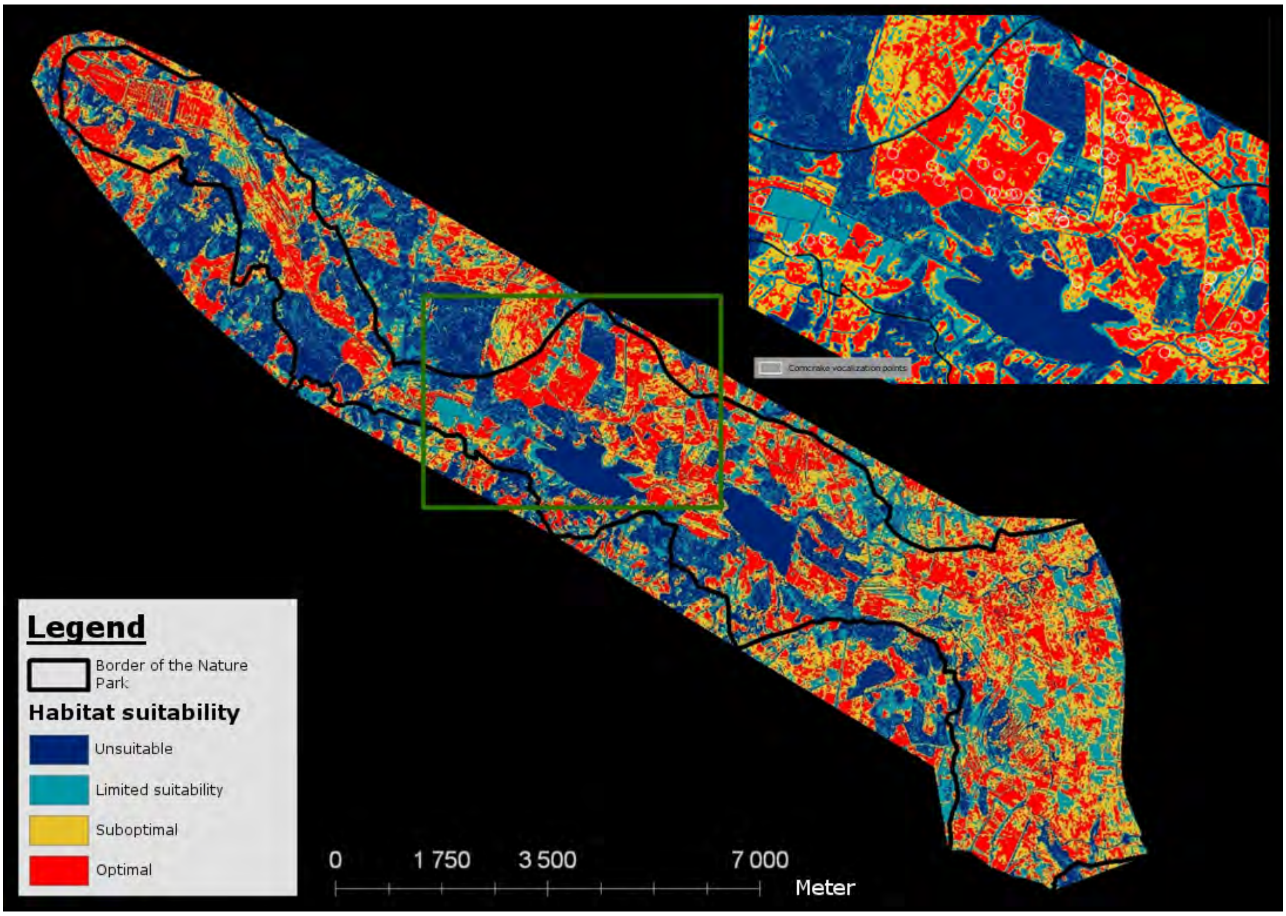


Figure 7. *Corncrake habitat suitability map in the Dviete floodplain Nature Park area in 2011.*

Figure 7 shows that grassland habitat suitability for Corncrakes in the Dviete floodplain Nature Park and its surrounding areas is high. This coincides with Latvia's Corncrake expert opinion that the Dviete floodplain Nature Park is a territory with one of the highest observed Corncrake densities in Latvia (Račinskis 2005). Comparatively high habitat suitability is also indicated by the 0.749 ENFA model species tolerance rating (value range 0–1) measured in *BioMapper*. A detailed description of the tolerance rating calculation can be found in A. H. Hirzel's (2004) work. If the rating is closer to 0, then the species' requirements are narrowly specific in the analysed territory (Hirzel 2004). The rating in this research is closer to 1, thereby indicating appropriate habitat conditions for the Corncrake in the Dviete floodplain Nature Park.

It is advisable to continue the removal of dense shrubs in the future, which has already been performed in the Skuķu Lake vicinity in 2011 as part of the LIFE+ project. To ascertain situational improvements, a repeat ENFA modelling would be necessary after the project's end.

To determine the origin of grassland fragments with limited habitat suitability shown in Figure 7, more detailed research is necessary for the part of the

Table 3.
The division of four classes in the ENFA model, corresponding habitat suitability value, average Corncrake occurrence density and average land area for one Corncrake male.

No.	Class	Corncrake habitat suitability, %	Average Corncrake density Corncrake male count/ha)		Average land area for 1 Corncrake male, ha	
			1. count	2. count	1. count	2. count
1	Optimal habitat	66 – 100 %	0,12	0,16	8,02 ± 1,38	6,20 ± 2,59
2	Suboptimal habitat	24 – 66 %	0,11	0,14	8,87 ± 3,39	7,10 ± 5,23
3	Limited suitability habitat	1 – 24 %	0,19	0,23	5,22 ± 3,12	4,42 ± 3,21
4	Unsuitable habitat	0 %	0	0	0	0

Dviete floodplain Nature Park that is adjacent to Daugava River. Possibly, its origin can be explained by the combination of specific agricultural land use and soil conditions in this area that is the most active agricultural area of the Dviete floodplain Nature Park (Račinskis 2005). These studies should be linked with hyperspectral data usage in order to provide comprehensive information that can be used for further ecological research. The widely applicable information for remote sensing remains not yet fully explored in hyperspectral data. This is substantiated by a constant stream of newly published research connected with usage of hyperspectral data in research (Kirkland et al. 2002; Haboudane et al. 2004; Cho, Skidmore 2006; Sobrino et al. 2012).

Conclusions

Comparing the DSS and ENFA models with the existing knowledge about Corncrake habitat requirements, the more believable and realistic results are shown by the ENFA model. The model was used to create a habitat suitability map for the Dviete floodplain Nature Park. The Corncrake occurrence density and the expected number of Corncrakes within a predefined territory corresponding to habitat suitability classes provides a basis for further, more detailed studies about Corncrake habitat requirements. Further research about the interaction and importance of influencing factors could possibly simplify the modelling methodology and create a sufficiently precise and realistic DSS modelling methodology.

In this research the high resolution remote sensing data ensured the successful use of the ENFA model. The information contained in hyperspectral data provided a relevant investment in ENFA models precision and a close reflection of reality in the developed map. Remote sensing data collection with airplanes equipped with appropriate sensors is recommended for habitat suitability modelling not only for the Corncrakes, but also other organisms and studies at a landscape scale. The acquired data can be used for nature conservation planning and wide scale landscape ecological process research.

The developed habitat suitability map can be practically used for the realisation of the current LIFE+ project 'Restoration of Corncrake habitats in Dviete floodplain Natura 2000 site', as well as further Corncrake conservation and research activities in the Dviete floodplain Nature Park. The developed ENFA modelling methodology provides an opportunity to survey Corncrake habitat suitability elsewhere in Latvia and in the wider Corncrake distribution area.

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References

Angelstam P., Mikusiński G., Rönnbäck B-I., Östman A., Lazdinis M., Roberge J-M., Arnberg W., Olsson J. 2003. Two-dimensional Gap Analysis: A Tool for Efficient Conservation Planning and Biodiversity Policy Implementation. – AMBIO, 32(8): 523–534.

Atsma G. 2006. Habitat selection by the Corncrake *Crex crex*: importance of ecotope distribution and landscape composition in river floodplains. – Report Environmental Science, 294, 47 pp.

Auniņš A. 2010. Eiropas Savienības nozīmes īpaši aizsargājamie biotopi Latvijā. Noteikšanas metodika. Latvijas Dabas fonds, Rīga, 335 lpp.

Ben-Dor E., Patkin K., Banin A., Karnieli A. 2002. Mapping of several soil properties using DAIS-7915 hyperspectral scanner data – a case study over clayey soils in Israel. – International Journal of Remote Sensing, 23(6): 1043–1062.

Berg Å., Gustafson T. 2007. Meadow management and occurrence of Corncrake *Crex crex*. – Agriculture Ecosystems and Environment, 120: 19–144.

Chase J.M. 2011. Ecological Niche Theory. – In: Scheiner S.M., Willing M.R. (Eds.) The Theory of Ecology. University of Chicago Press, 404 pp.

Cho M.A., Skidmore A.K. 2006. A new technique for extracting the red edge position from hyperspectral data: the linear extrapolation method. – Remote Sensing of Environment, 101: 181–193.

Fattorini S. 2005. A simple method to fit geometric series and broken stick models in community ecology and island biogeography. – Acta Oecologica, 28: 199–205.

Fava F., Colombo R., Bocchi S., Meroni M., Sitzia M., Fois N., Zucca C. 2009. Identification of hyperspectral vegetation indices for Mediterranean pasture characterization. – International Journal of Applied Earth Observation and Geoinformation, 11(4): 233–243.

Gaucherel C., Martinet V., Bamiere L., Sheeren D., Gibon A., Joannon A., Castellazzi M.S., Boussard H., Barraquand F., Inchausti P., Lazrak E.G., Mari J.-F., Schaller N., Houet T., Bretagnolle V. 2010. A multidisciplinary modelling approach to analyse and predict the effects of landscape Dynamics on biodiversity. – In: International Conference on Integrative Landscape Modelling „LANDMOD2010", 3–5 February, 2010, Montpellier, 10 pp.

Grabovsky V.I. 1993. Spatial distribution and spacing behaviour of males in a Russian Corncrake (Crex crex) population. – Gibier Faune Sauvage 10, 259–279.

Green, R. E. 1996. Factors affecting the population density of the Corncrake *Crex crex* in Britain and Ireland. – Journal of Applied Ecology: 237–248.

Green R.E., Tyler G.A., Stowe T.J., Newton A.V. 1997. A simulation model of the effect of mowing of agricultural grassland on the breeding success of the Corncrake (Crex crex). – Journal of Zoology, 243(1): 81–115.

Haboudane D., Miller J.R., Pattey E., Zarco-Tejada P.J., Strachan I. B. 2004. Hyperspectral vegetation indices and novel algorithms for predicting green LAI of crop canopies: Modeling and validation in the context of precision agriculture. – Remote Sensing of Environment, 90(3): 337–352.

Hirzel A.H., Hausser J., Chessel D., Perrin N. 2002. Ecological-niche factor analysis: how to compute habitat-suitability maps without absence data? – Ecology, 83(7): 2027–2036.

Hirzel A.H. 2004. BioMapper 3. User's manual. http://www2.unil.ch/biomapper/Biomapper3-Manual.pdf

Hirzel A.H., Gay G.L., Helfer V., Randin C., Guisan A. 2006. Evaluating the ability of habitat suitability models to predict species presences. – Ecological Modelling, 199: 142–152.

Jooss R., Geissler-Strobel S., Trautner J., Hermann G., Kaule G. 2009. 'Conservation responsibilities' of municipalities for target species Prioritizing conservation by assigning responsibilities to municipalities in Baden-uerttemberg, Germany. Landscape and Urban Planning, 93: 218–228.

Keišs O. 2005. Impact of changes in agricultural land use on Corncrake *Crex crex* population in Latvia. –Acta Universitatis Latviensis, Biology, 691: 93–109.

Keišs O. 2006. Lauksaimniecības pārmaiņu ietekme uz griezes *Crex crex* (L.) populāciju Latvijā: skaita dinamika, biotopu izvēle un populācijas struktūra. Disertācija bioloģijas doktora grāda iegūšanai. Latvijas Universitāte, Bioloģijas fakultāte, Rīga, 100 lpp.

Ķerus V., Reihmanis J., Drzdovskis D., Avotiņš A. jun. 2014. Griezes *Crex crex* un citu lauksaimniecības zemēs sastopamo naktsputnu skaita pārmaiņas Dvietes dabas parkā no 2006. Līdz 2014. gadam. Latvijas Ornitoloģijas biedrība, Rīga, 16 lpp.

Kirkland L., Herr K., Keim E., Adams P., Salisbury J., Hackwell J., Treiman A. 2002. First Use of an Airborne Thermal Infrared Hyperspectral Scanner for Compositional Mapping. – Remote Sensing of Environment, 80(3): 447–459.

Lu D., Weng Q. 2007. A survey of image classification methods and techniques for improving classification performance. – International Journal of Remote Sensing, 28(5): 823–870.

Ministru kabineta noteikumi Nr. 295 „Noteikumi par valsts un Eiropas Savienības lauku attīstības atbalsta piešķiršanu, administrēšanu un uzraudzību vides un lauku ainavas uzlabošanai". Spēkā esoši kopš 2010. gada 31. marta. Pēdējie grozījumi stājas spēkā 2014. gada 28. martā.

Numata I., Roberts D.A., Chadwick O.A., Schimel J., Sampaio F., Leonidas F., Soares J. 2007. Characterization of pasture biophysical properties and the impact of grazing intensity using remotely sensed data. – Remote Sensing of Environment 109(3): 314–327.

Opdam P., Foppen R., Vos C. 2002. Bringing the gap between ecology and spatial planning in landscape ecology. – Landscape Ecology, 16: 767–779.

Ortega-Huerta M.A., Peterson A.T. 2008. Modeling ecological niches and predicting geographical distribution: a test of six presence-only methods. – Revista Mexicana de Biodiversidad, 79: 205–216.

Osborne P.E., Alonso J.C., Bryant R.G. 2001. Modelling landscape-scale habitat use using GIS and remote sensing: a case study with great bustard. – Journal of Applied Ecology, 38: 458–471.

Račinskis E. (red.) 2005. Dabas parka Dvietes paliene dabas aizsardzības plāns. Latvijas Ornitoloģijas biedrība, Rīga, 73 lpp.

Reihmanis J. 2011. Riestojošo griežu *Crex crex* telpiskais izvietojums un riesta teritoriju analīze. Autorlīguma par pasūtīto darbu nr. VRI 25-02/4/2 atskaite. 8 lpp.

Sanderson E.W., Redford K.H., Vedder A., Coppolillo P.B., Ward S.E. 2002. A conceptual model for conservation planning based on landscape species requirements. – Landscape and Urban Planning, 58: 41–56.

Schäffer N., Koffijberg K. 2004. *Crex crex* Corncrake. Handbook of the Birds of Europe the Middle East and North Africa. Birds of the Western Palearctic, 6(1–2): 55–76.

Skliba J., Fuch R. 2004. Male Corncrakes *Crex crex* extend their home ranges by visiting the territories of neighbouring males: Capsule Radiotracked male Corncrake often intruded on the territories of neighbouring males. – Bird Study, 50(2): 113–118.

Sobrino J.A., Franch B., Mattar C., Jiménez-Muñoz, Corbari C. 2012. A method to estimate soil moisture from Airborne Hyperspectral Scanner (AHS) and ASTER data: Application to SEN2FLEX and SEN3EXP campaigns. – Remote Sensing of Environment, 117: 415–428.

Sokal R.R., Rohlf F.J. 1995. Biometry: The principles and practice of statistics in biological research. 3rd ed. W.H.Freeman, New York, 887 p.

Store R., Kangas J. 2001. Integrating spatial multi-criteria evaluation and expert knowledge for GIS-based habitat suitability modelling. – Landscape and Urban Planning, 55: 79–93.

Stowe T.J., Hudson A.V. 1991. Radio-telemetry studies of Corncrake in Great Britain. – Vogelwelt, 112: 10–16.

Strubbe D., Matthysen E. 2008. Predicting the potential distribution of invasive ring-necked parakeets Psittacula krameri in northern Belgium using an ecological niche modelling approach. – Biological Invasions 11 (3): 497–513.

Suchant R., Baritz R., Braunisch V. 2003. Wildlife habitat analysis – a multidimensional habitat management model. – Journal of Nature Conservation, 10: 253–268.

Thenkabail P. S., Lyon J. G., & Huete, A. (Eds.). 2012. Hyperspectral remote sensing of vegetation. New York, CRC Press, 781 pp.

Uran O., Janssen R. 2003. Why are spatial decision support systems not used? Some experiences from the Netherlands. – Computers, Environment and Urban Systems, 27: 511–526.

van Weperen M. 2009. Habitat selection of the Corncrake (Crex crex) in floodplains along the Dutch Rhine River branches. Radboud Universiteit Nijmegen, 65 pp.

Wettstein W., Szep T., Kery M. 2001. Habitat selection of Corncrakes (Crex crex L.) in Szatmar-Bereg (Hungary) and implications for further monitoring. – Ornis Hungarica, 11: 9–18.

Wulder M. A., Hall R. J., Coops N. C., Franklin S. E. 2004. High spatial resolution remotely sensed data for ecosystem characterization. – BioScience, 54(6): 511–521.

Changes in Vegetation Resulting from Management in the Lielupe Floodplain Grassland Area in Ķemeri National Park

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Summary

The aim of this paper was to analyze changes in grassland vegetation during a 12-year period (from 2003 to 2014) in the Lielupe floodplain grassland complex in Ķemeri National Park, Latvia. Vegetation changes have been the result of mowing, year-round grazing using Heck cattle and Konik horses, paludification and other factors. The results of vegetation monitoring suggest that the changes are not linear, the influencing factors (mowing, grazing, hydrological changes) are mutually related, and the inter-relation among these impacts is difficult to untangle. During the monitoring period the changes in plant species composition in the unmanaged sectors of grassland area were not significant. At the same time, the most significant vegetational changes occurred in some abandoned parts of the grasslands affected by beaver-caused paludification, which caused complete turn-over of species in some parts of the area. Thus, in this particular case, paludification was a more significant factor causing changes in vegetation than mowing and grazing.

Introduction

In contemporary Europe, alluvial grasslands are among the most threatened ecosystems, at the same time being rich in different species and communities and performing several important ecosystem services (Tockner et al. 2008; Krause et al. 2011). During the 20th century, alluvial grasslands were significantly altered by drainage, change of land use and other human-caused modifications, especially during the second half of the 20th century.

Most floodplains in Latvia have been modified by drainage and cultivation. In the 1990s, many alluvial grasslands, which were previously used for hay cutting and grazing, were abandoned due to socio-economic reasons. Some grasslands were abandoned earlier, particularly those in remote areas with limited access. According to a report on the conservation status of species and habitats of European Union importance in Latvia (2013), the total area of alluvial grasslands (habitat type 6450 *Northern boreal alluvial meadows*) is estimated at 156 km^2. The conservation status is evaluated as unfavorable. Hydrological modification and cessation of regular management are considered the main causes for the degradation (Anon. 2013).

The aim of this paper is to assess the success of the restoration and subsequent management of alluvial semi-natural grasslands in the Lielupe grassland area in Ķemeri National Park from 2003 to 2014.

Material and methods

Study area

The study area (ca. 280 ha) lies in the Coastal Lowland, on the south-east edge of Ķemeri National Park, on the left bank of the Lielupe River (Fig. 1). The territory is approximately 0.2 to 1.4 m above the sea level. The terrain is flat, with topographic variations typical for floodplains, where the micro-elevations do not exceed one meter. The entire territory is seasonally flooded. Alluvial soils prevail, peat soils are found in wet depressions.

During the 20th century, the entire area was drained several times (Fig. 2). Some of the oldest ditches were dug in the beginning of the 20th century (Liepa, Ķuze 2004), while the majority of the present-day ditches were dug around the mid-20th century. When digging the ditches, in most cases embankments were made on one or both sides of the ditch disposing the excavated soil. The embankments hinder the natural

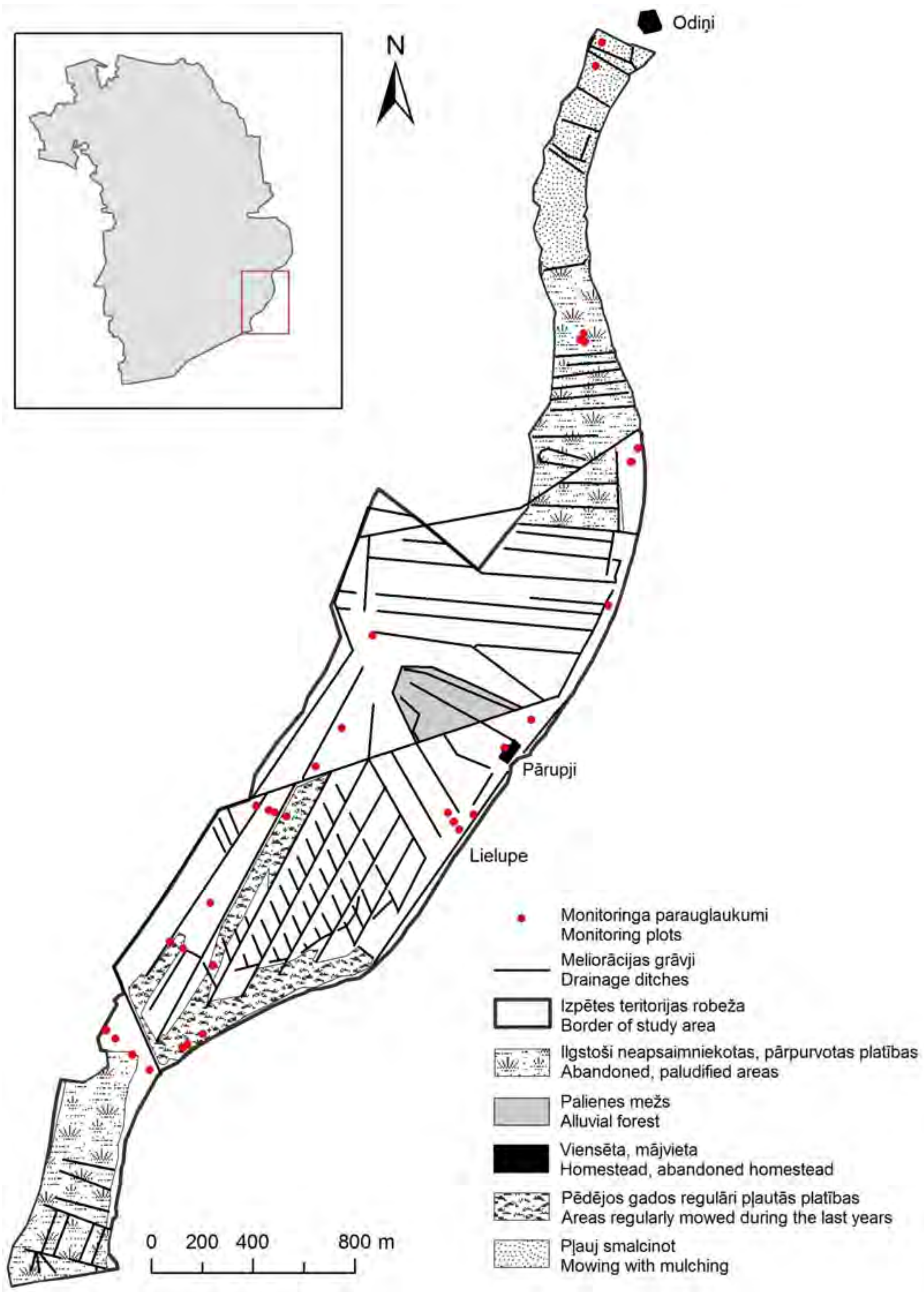
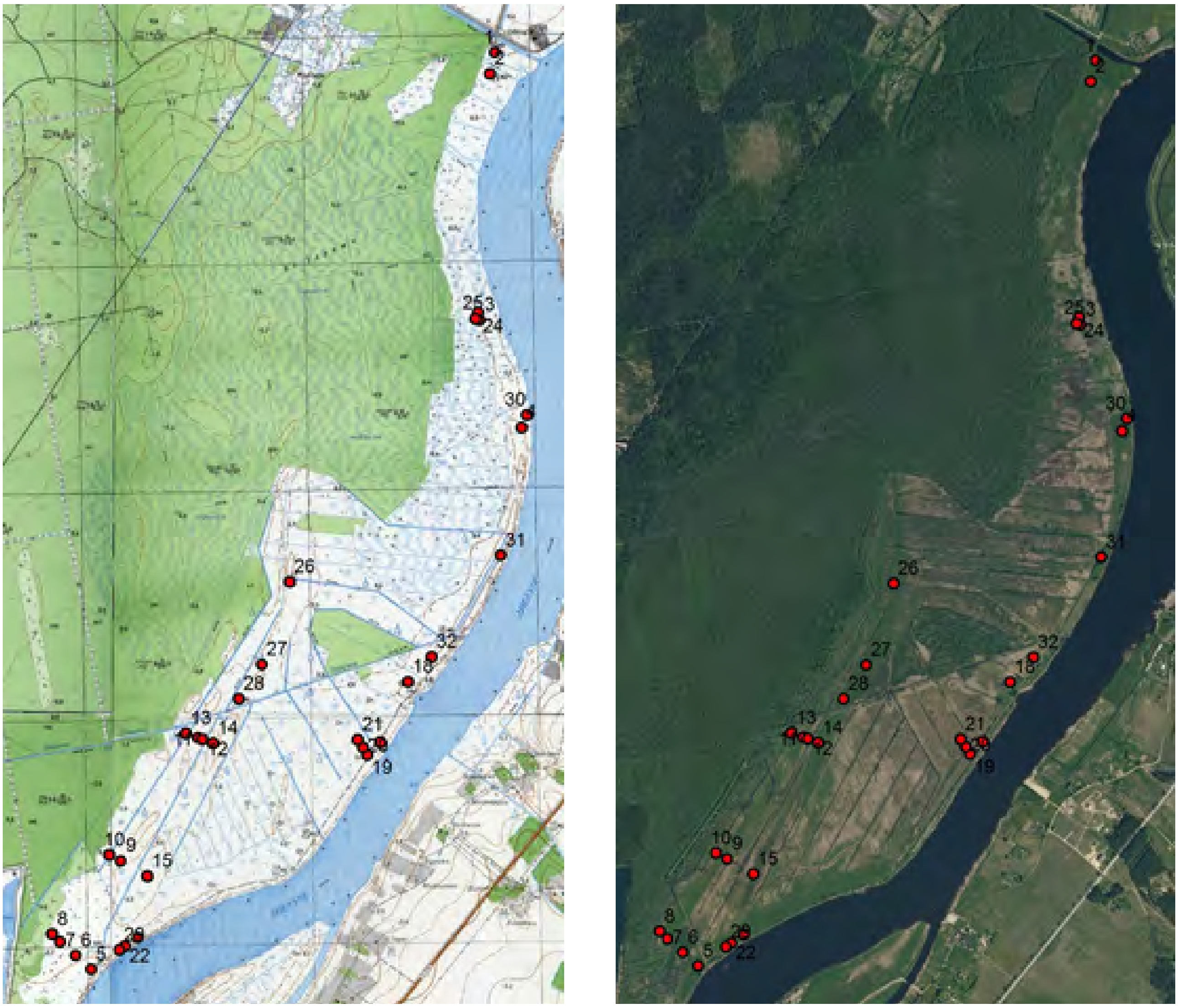


Fig. 1. *Map of the study area showing the main management practices and paludified areas. Most areas are accessible to cattle and horses.*

flow of water and are suitable for beavers to establish their dams. In many cases, the embankments are overgrown with trees and shrubs (Liepa, Ķuze 2004). In the central part of the area near the former Pārupji homestead, the bank of the Lielupe River was strongly modified by digging a wide canal between the river and the grassland area and building a dike. The dike partly functions as a barrier during the flood season, though during the high water season the entire area gets flooded.

Until World War II, there was a homestead called Pārupji in the central part of the grassland area on a slight elevation. The surroundings of the homestead were probably used as pasture. However, from the mid-20th century to the 1990s the entire area was used for hay cutting. As suggested by some aerial photos taken in the 1940s, there were several sheds which were used for storing the hay. In 2004, the hay cutting was done in only 10–15 ha, while most of the area was abandoned.



20. gs. 60. gadi / 1960s

2007. gads / 2007

Fig. 2. *The drainage system of the Lielupe grassland area in the 1960s (on the left) and in 2007 (on the right). The vegetation monitoring plots are marked with red dots. The background: topographic map of general headquarters of the Latvian SSR and orto-photo map (Latvian Geospatial Information Agency).*

The land is owned by the state (Ministry of Environmental Protection and Regional Development) (66 %), and by private owners. All of the state-owned land and some of the privately owned land is rented out to the non-governmental organization Ķemeri National Park Fund, which is responsible for the restoration and management of the grasslands. In recent years, the mowed areas have increased to 30–35 ha. Some sectors are fenced off in the spring and used for hay making. In late autumn the area is again made available to the animals. Mowing with mulching is applied in the northern part of the area to the south of the Odiņi homestead and in some sectors to the south of Pārupji. Since 2006, approximately 2/3 of the area is used for grazing year-round by *Heck* cattle and *Konik* horses. In 2006, 32 horses and five cows were brought to the area. The number of animals has grown to 36 horses and 29 cows in 2014, excluding colts and calves. In most of the area, the grasslands were managed using the agri-environmental support payments for biologically valuable grasslands within the Rural Development Programme.

Vegetation monitoring methods

The data from vegetation monitoring from 2003 to 2014 are used in this paper. The monitoring was started in 2003 by establishing 22 circle monitoring plots with a 2 m diameter. The data were collected each year except 2011. In 2007 ten additional monitoring plots were established. The coordinates for the centers of each monitoring plot were recorded using a high precision Trimble GPS device (precision ±0.5 m). The centers of the monitoring plots were also marked using low wooden poles so as not to hinder mowing. Finding the wooden poles sometimes was difficult and in some cases might have led to errors in determining the center of some monitoring plots. In 2007–2008 the wooden poles were replaced by metal ones so that a metal detector could be used in addition to GPS for finding the plots.

During each survey all vascular plant species were recorded for each plot and their cover was estimated visually as a percentage using the Braun-Blanquet method (Braun-Blanquet 1965). All impacts that could lead to changes in vegetation, e. g. intensity of grazing and trampling, signs of beaver activity etc., were also recorded. Since 2007 photos of each plot were taken.
In 2007 the vegetation mapping was carried out in the area using national habitat classification (Kabucis 2001).

The monitoring was carried out by E. Grolle from 2003–2005, E. Biseniece in 2006, A. Priede from 2007–2011 and A. Priede and V. Caune in 2013–2014. The change in personnel doing the monitoring was taken into account when analyzing the data.

The monitoring was carried out in an ongoing and dynamic habitat restoration and the management situation preventing the establishment of control plots. In most cases the plots were established several years before the beginning of the restoration and management activities in that specific area of the grassland. There were some influencing factors that could not be predicted, for example the activity of beavers. The spatial distribution and intensity of grazing also changed from year to year. That led to a different number of plots in each management group when grouping the plots according to the management activities for data analysis.

Data analysis

The vegetation data were stored in a data base using the vegetation database software TURBOVEG 2.3 (Hennekens, Schaminee 2001). A detrended correspondence analysis (DCA) was performed using Canoco 4.5 software (Ter Braak, Smilauer 2002) to evaluate the effect of different management practices on vegetation. The analysis was carried out using data on relative species cover from 2008 and 2013 using square root transformation and downweighting of rare species. The data sets from 2008 and 2013 were chosen because it allowed the use of the largest number of samples and at the same time prevented some errors that could arise from changes in personnel performing the monitoring or inaccuracies in finding the centers of monitoring plots. The ordination diagrams for both plots and species were created. The plots were grouped according to the main management method from 2008–2013.

Several monitoring plots were excluded from the analysis. Plot No 6 was excluded because from 2010 it was paludified and no longer accessible, therefore the monitoring was stopped. Plot No 15 was excluded because from 2009–2010 it had a hay stack sitting on it. Plot No18 was excluded after the initial data analysis, because it had a very different species composition (dominated by *Phalaroides arundinacea*) in comparison with other plots and it made the ordination diagram unreadable.

A separate chart of changes in species cover and composition was created for each monitoring plot. It was done to gain more detailed information on the nature of vegetation changes and to help explain the results of DCA analysis.

The species in these charts were grouped based on similarities in their ecology. The groups were established based on the descriptions of protected grassland habitats of EU importance in Latvia (Rūsiņa 2010): **wet grassland species** (tall sedges) – *Carex acuta, C. acutiformis, C. cespitosa, C. disticha, C. nigra, C. riparia, C. vesicaria, C. vulpina, Glyceria maxima, Phalaroides arundinacea;* **moderately moist (mesophilous) to moist grassland species** – *Agrostis gigantea, Alopecurus pratensis, Briza media, Campanula patula, Centaurea jacea, Dactylis glomereata, Festuca pratensis, Galium album, G. boreale, Geranium palustre, Geum rivale, Helictotrichon pubescens, Heracleum sibiricum Lathyrus pratensis, Lychnis flos-cuculi, Phleum pratense, Plantago media, Poa palustris, Ranunculus auricomus, Stellaria graminea, Tragopogon pratensis, Trisetum flavescens, Vicia cracca;* **pasture species** – *Alchemilla vulgaris, Anthoxantum odoratum, Briza media, Caltha palustris, Carex panicea, Crepis paludosa, Deschampsia cespitosa, Festuca ovina, F. rubra, Galium uliginosum, Geum rivale, Holcus lanatus, Luzula multiflora, Plantago lanceolata, P. media, Potentilla erecta, Primula veris, Prunella vulgaris, Trifolium repens, Trollius europaeus;* **overgrowth indicators for wet grasslands** – *Filipendula ulmaria, Phragmites australis;* **overgrowth indicators for moderately moist to moist grasslands** – *Anthriscus sylvestris, Cirsium arvense, Salix cinerea, Urtica dioica, Elytrigia repens;* **trampling indicators** – *Plantago major, Poa annua, Polygonum aviculare, Potentilla anserina, Ranunculus repens, Taraxacum officinale, Trifolium repens*.

The changes in species composition and cover in each monitoring plot were visualized using MS Excel.

Results and discussion

Vegetation types and their changes at landscape level

According to the vegetation mapping results from 2007, the largest areas in the Lielupe floodplain grassland complex were covered by tall sedge communities (ca. 34 %) dominated mostly by *Carex disticha* and *Carex acuta*. 32 % of the area was covered by reed beds, the former tall sedge communities, which occur in the wettest parts of the area and result from abandonment since the mid-20th century. Moderately moist grasslands dominated by *Festuca pratensis*, *Centaurea jacea* etc. occupied ca. 17 % of the area, *Glyceria maxima* dominated wet grasslands – ca. 3 %, hydrophilous tall herb fringe communities – ca. 2 %, other vegetation types (ruderal communities around Pārupji homestead, paths etc.) – ca. 1 %. 11 % of the area is covered by shrubs, mostly willow stands, which occur both on drier and wetter parts of the grassland area.

Although no detailed mapping was carried out repeatedly, in 2014 the situation had changed considerably. The areas covered by shrubs had decreased (partly removed), the areas of reed-dominated grassland patches had also decreased. The most significant changes related to management (mowing and grazing) were found in the intensively grazed sectors of the area, where the horses and cattle stay for most of time, especially during the summer – along the bank of the Lielupe River. Within 3–4 years since the establishment of the pasture, the former reed beds have transformed into open pasture with hummocks and low sward due to intensive grazing. During the autumn and winter periods, the grazing animals visit the sectors with low grazing intensity during the summer, also browsing the shrubs. However, since the animal density per hectare is still relatively low, considerable areas in the wettest parts of the floodplain with limited access are still overgrown with shrubs and reeds, which continue their expansion.

Since the establishment of the pasture, significant changes have also occurred on the bank of Lielupe River. It was dominated by hydrophilous tall herb vegetation (*Calystegietalia*) and reeds, but in 2014 many patches were grazed and transformed into short sward vegetation.

In the southern part of the area, the water table has increased causing hydrological changes in several hectares due to the beaver activity. As a result of paludification, the formerly moderately moist to moist grassland has transformed into tall sedge communities, while the former tall sedge communities have turned into eutrophic freshwater plant communities with *Typha latifolia* and *Phragmites australis*.

Vegetation changes in monitoring plots

In the ordination diagram of the DCA ordination of monitoring plots (Fig. 3) the plots, which showed signs of paludification are grouped to the right. In the DCA species ordination diagram (Fig. 4), the most significant species in these plots were *Phragmites australis* and *Carex cespitosa*. The plots with moderately moist grassland vegetation are grouped to the left. So it is likely that the first ordination axis is related to moisture.

When looking at the changes in vegetation from 2008–2013 in the ordination diagram one can see that the plots, which were grazed and one plot that was paludified show the largest changes. The monitoring plots in moderately moist grasslands, which were moved with gathering of hay and grazed in aftergrass as well as some abandoned plots show very little or no change. Relatively moderate change can be seen in the plots that were mowed with mulching and in some of the abandoned plots.

The plots that were situated in wet grasslands show more change that those that were situated in moderately moist grasslands.

Perhaps some of the differences in the levels of change between mowed and grazed grasslands can be explained by their management in the past. Historically the Lielupe floodplain meadows were used for hay making. Before these grasslands were abandoned their species composition and vegetation structure was that of hay meadows. When the grazing started, it initiated changes from hay meadow vegetation to that of a pasture. It can be clearly seen in the examples of individual monitoring plots, especially the ones that were grazed intensively, e. g. plot No 31 (Fig. 5). In just a few years the cover of mesophilous grassland species was severely reduced and they were replaced by species that are accustomed to grazing and heavy trampling. The resulting vegetation has a lawn-like appearance. This example also shows that just after the grazing started and the grazing pressure was low, the cover of mesophile grassland species actually increased. It must be noted that in the Lielupe floodplain meadows very few areas are so intensively grazed.

In the plots, which were grazed less intensively the species turnover was more gradual. It is hard to predict what the end result will be as the observation period – 12 years, is still too short. Research in other countries shows that changes in species composition in pasture grasslands that are restored using mowing still continues after 20 years (Poptcheva et al. 2009). Wet grasslands that are moderately grazed also show a decrease in wet grassland species and overgrowth indicators and an increase of species that tolerate trampling. Interestingly the cover of species that

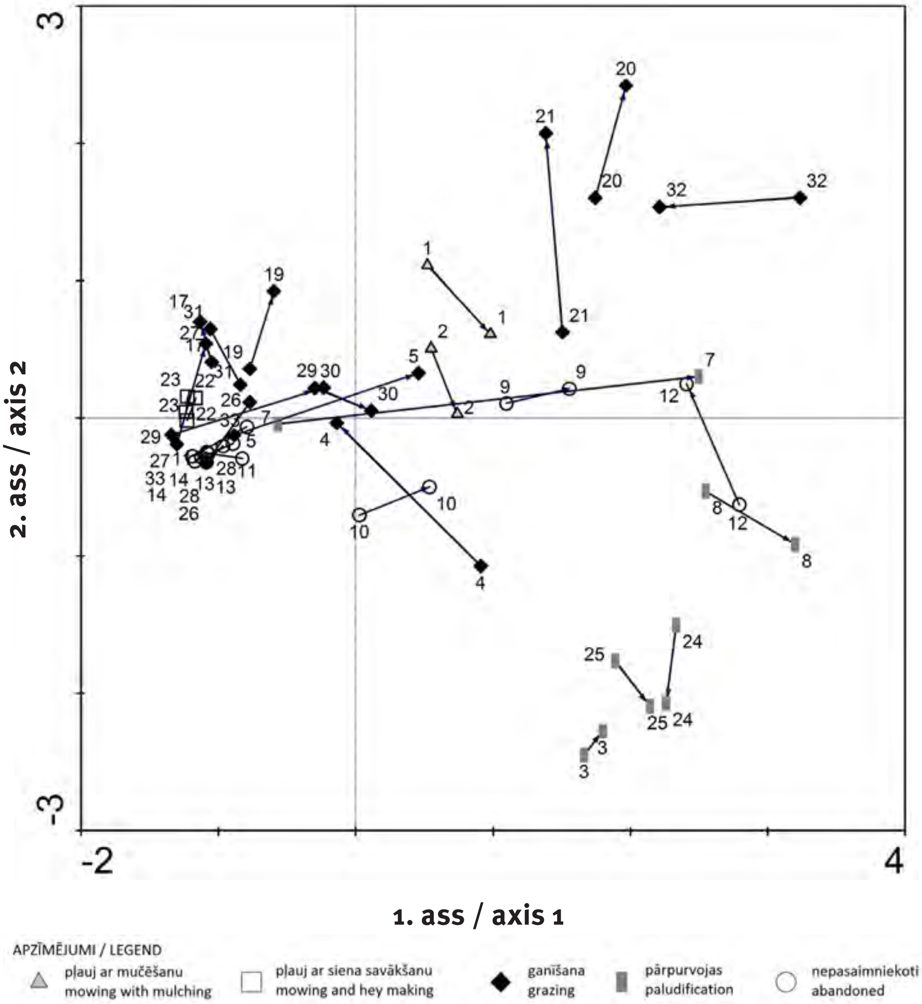


Fig. 3. *DCA ordination diagram of vegetation monitoring plots in the Lielupe floodplain meadows. Arrows connect each plot in the ordination space in 2008 and 2013. The plot numbers are the same as used in the monitoring.*

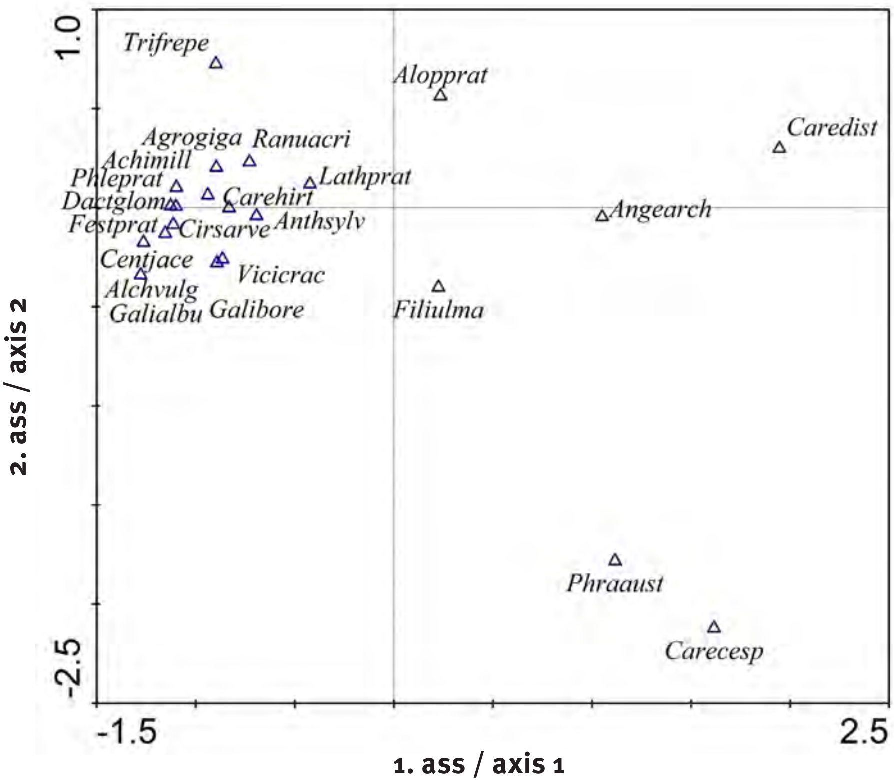


Fig. 4. *DCA species ordination diagram for vegetation monitoring plots in the Lielupe floodplain meadows. In the species names, abbreviations (the first four letters of the name of genus and species) are used:* Trifrepe – *Trifolium repens*, Alopprat – *Alopecurus pratensis*, Caredist – *Carex disticha*, Angearch – *Angelica archangelica*, Phraaust – *Phragmites australis*, Carecesp – *Carex cespitosa*, Filiulma – *Filipendula ulmaria*, Agrogiga – *Agrostis gigantea*, Ranuacri – *Ranunculus acris*, Achimill – *Achillea millefolium*, Lathprat – *Lathyrus pratensis*, Phleuprat – *Phleum pratensis*, Carehirt – *Carex hirta*, Dactglom – *Dactylis glomerata*, Festprat – *Festuca pratensis*, Cirsarve – *Cirsium arvensis*, Anthsylv – *Anthriscus sylvestris*, Centjace – *Centaurea jacea*, Vicicrac – *Vicia cracca*, Alchvulg – *Alchemilla vulgaris*, Galialb – *Galium album*, Galibore – *Galium boreale*.

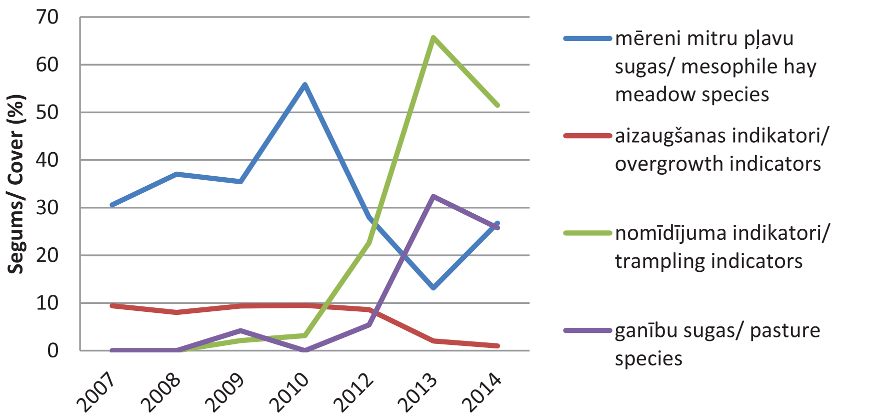


Fig. 5. *Changes in the relative cover of species ecological groups in mesophile grassland that has been intensively grazed since 2009 (example: sample plot No. 31).*

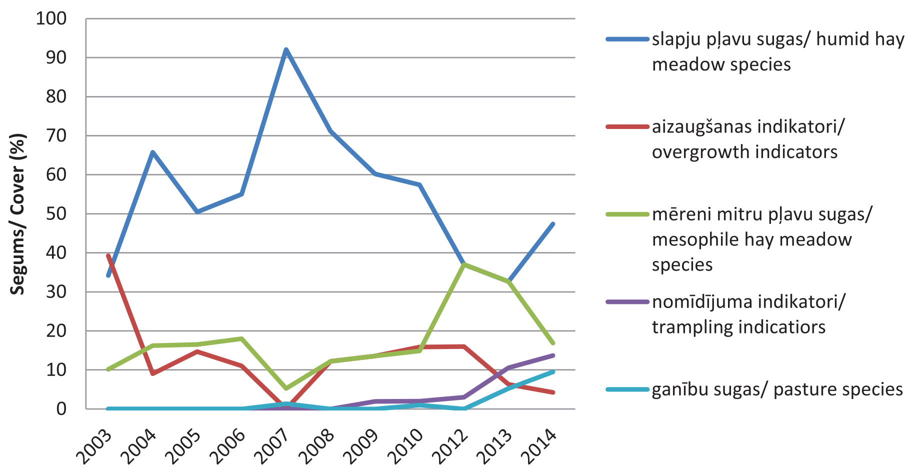


Fig. 6. *Changes in the relative cover of species ecological groups in grazed humid grassland (example: sample plot No. 4).*

are characteristic to moderately moist to moist hay meadows and pastures also slightly increases (Fig. 6). This means that the grazing reduces the amount of the accumulated dead grass and more moisture can escape through evaporation making the grassland slightly drier. The grazing also improves the light conditions in the lower strata of the vegetation. The tall sedges and reed are replaced by a mosaic of short vegetation and tussocks. It creates favorable conditions for a number of moderately moist, moist and wet grassland species (Fig. 7).

The fact that the changes in the grazed monitoring plots in the ordination diagram go in various directions can be partially explained by the uneven impact of grazing in comparison to mowing. Some spots are grazed moderately, some heavily, some are just trampled. This is closely connected to the behavior of grazing animals, which in turn is influenced by a myriad of factors. According to other authors, some of the factors that determine the spatial differences in grazing intensity are the forage quality of different grassland species (Poaceae is preferred over Cyperaceae), vegetation structure (height etc.), availability of water sources, sources of minerals and resting areas (Schaich et. al. 2010).

The monitoring plots in moderately moist to moist grasslands that are mowed for hay and grazed in aftergrass show a slight decrease in mesophile grassland species and a slight increase in pasture species, but no increase in species tolerant to trampling (Fig. 8). The areas are fenced off in the beginning of summer and made available to grazing animals only in late autumn. It prevents the compacting of the soil and the creation of heavily trampled areas or areas with bare soil.

The relationships among the ecological groups of species in abandoned moderately moist to moist grasslands show relatively little change during a 12 year period (Fig. 9). The cover of overgrowth indicators has increased by 10 % indicating that the overgrowth process is very slow.

Some monitoring plots that were considered abandoned showed a slight increase of pasture species (Fig. 9), though during monitoring no signs of grazing were observed. They are probably grazed in late autumn and winter when animals are forced to travel further to find food. Such seasonal changes in grazing intensity are observed in other studies as well (Schaich et al. 2010).

As can be seen in the plot ordination diagram (Fig. 10), the changes in abandoned wet grasslands occur more rapidly than in moderately moist ones. In most cases the tall sedge communities overgrow with *Phragmites australis*. In some cases also with *Typha latifolia* or *Glyceria maxima*.

Another explanation for the fact that vegetation changes in the monitoring plots situated in abandoned grasslands are slow could be related to the spatial position of these monitoring plots. In many cases the overgrowth of grassland patches starts at the edges and progresses towards the center. In the monitoring plots situated closer to the edges of the grassland the overgrowth process will be observed sooner, giving the impression that it is more rapid.

In the monitoring plots that are paludified due to beaver activity, the vegetation changes are quite radical. For example the vegetation in plot No 7 changed from mesophilous grassland to wet grassland, dominated by sedges, in just 2–3 years (Fig. 11 and 12).

At the same time, in a nearby plot (No 8) wet grassland vegetation dominated by tall sedges (plot No 8) was replaced by species characteristic of paludified areas, mostly Typha latifolia (Fig. 13).

Scientific publications on the changes of vegetation during the paludification of floodplain grasslands and its effects from the point of view of biodiversity conservation could not be found. We consider paludification can cause dramatic changes in grassland vegetation. It also severely hinders grassland management. Beaver activity in the floodplains should be monitored and the beaver dams should be destroyed. The systematic destruction of beaver dams in the study area was started in 2013. At this point in time it is not yet possible to evaluate the speed at

2007

2013

Fig. 7. *Changes in the vegetation structure in grazed humid grassland (example: sample plot No. 4). Photos: A. Liepa.*

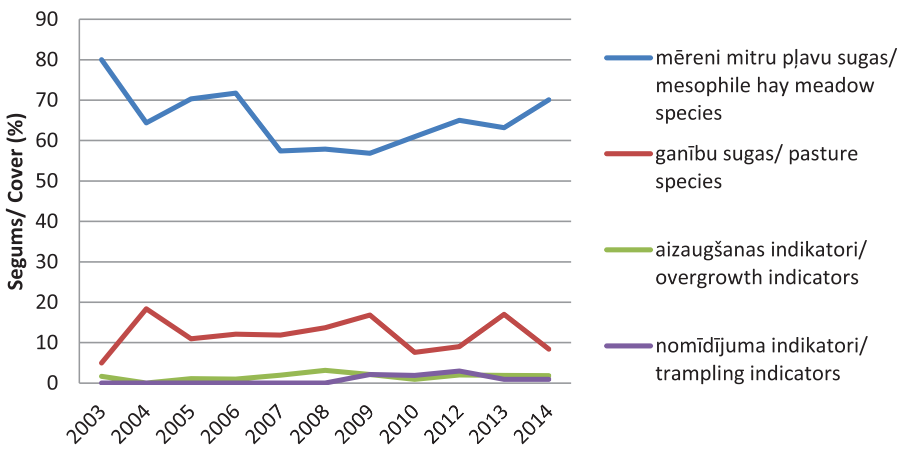


Fig. 8. *Changes in the relative cover of species ecological groups in mesophile grassland that has been managed by mowing followed by grazing in aftergrass (example: sample plot No. 14).*

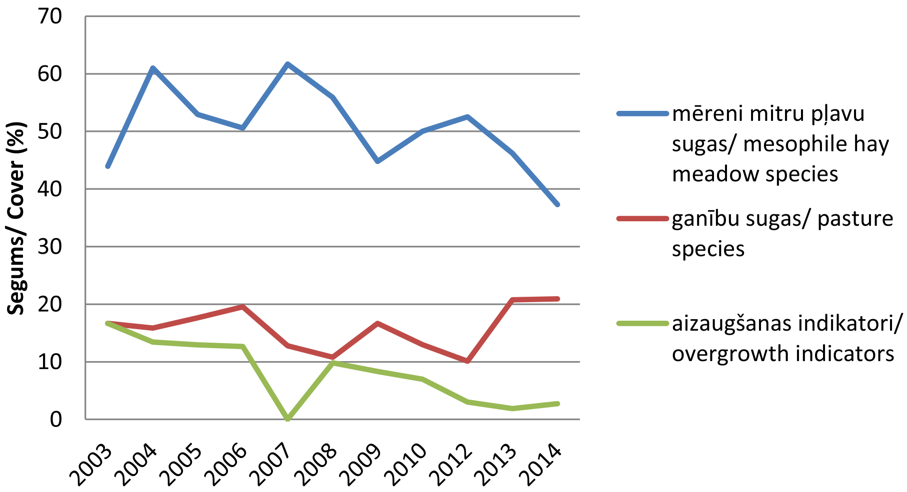


Fig. 9. *Changes in the relative cover of species ecological groups in abandoned mesophile grassland with a relatively stable water table (example: sample plot No. 11).*

2008

2013

Fig. 10. *Abandoned humid grassland overgrown with the common reed Phragmites australis (example: surroundings of the sample plot No. 3). Photos: A. Liepa.*

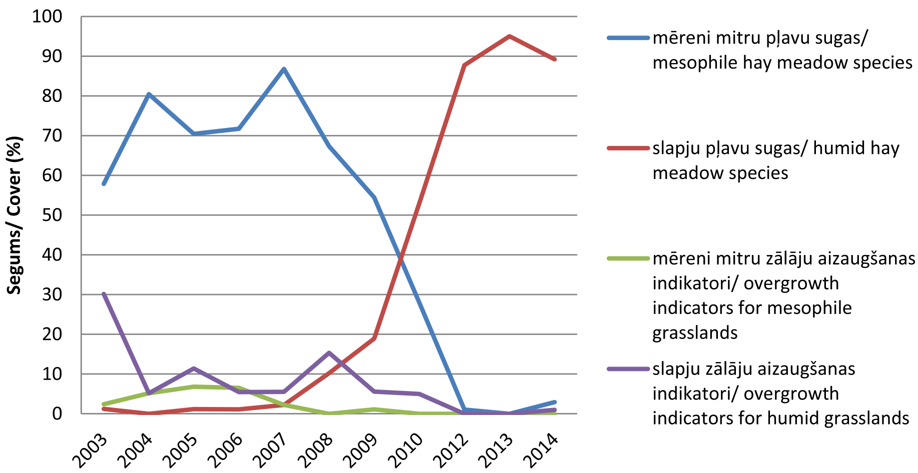


Fig. 11. *Changes in the relative cover of species ecological groups in abandoned mesophile grassland flooded by beaver (example: sample plot No. 7).*

2007

2013

Fig. 12. *Changes in vegetation due to paludification (example: sample plot No. 7). Photos: A. Liepa and A. Priede.*

2007

2013

Fig. 13. *Changes in vegetation due to paludification (example: sample plot No. 8). The tall sedge communities are being replaced by pioneer communities typical for shallow euthrophic waters. Photos: A. Liepa and A. Priede.*

which the grassland vegetation will recover and how exactly it will happen. It is likely that the former tall sedge communities will recover relatively quickly – in 2–3 years once the dryer conditions are restored. On the other hand the changes in the moderately moist grasslands that have been replaced by tall sedge communities might take more time and be more non-linear as the sedges are strong competitors and often form mono-dominant vegetation.

Conclusions

The restoration and management of the Lielupe floodplain grasslands using year-round grazing is related to changes at both the landscape and plant community level. At the landscape level the total area of open grassland with low vegetation and a mosaic structure has increased. In the vegetation monitoring plots previously dominated by hay meadow species and some overgrowth indicators, the cover of pasture related species has increased, especially in heavily grazed areas.

The areas that are managed by mowing, hay making and grazing in aftergrass have not changed much at both the landscape and plant community level. Most probably because mowing and hay making was the management method that was used in the area in the past.

The total area of grassland habitats lost to overgrowth in parts of the grassland complex that is still abandoned has not been assessed. At the level of individual monitoring plots the overgrowth process seems to happen relatively slowly, especially in mesophilous grasslands. But these results can also be related to the positioning of the monitoring plots.

Paludification due to beaver activity causes the most severe and undesired changes in grassland vegetation. In 2–3 years the vegetation undergoes radical changes at both the landscape and plant community level. It is too early to estimate how fast or successful the recovery process will be after the beaver dams were destroyed in 2013.

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References

Anon. 2013. Conservation status of species and habitats. Reporting under Article 17 of the Habitats Directive. Latvia, assessment 2007–2012 (2013), European Commission, http://cdr.eionet.europa.eu/lv/eu/art17/envuc1kdw

Braun-Blanquet J. 1965. Plant sociology: The study of plant communities. Hafner, London.

Hennekens S. M., Schaminee J. H. J. 2001. Turboveg, a comprehensive database management system for vegetation data. – Journal of Vegetation Science 12: 589–591.

Kabucis I. (red.). 2001. Latvijas biotopi. Klasifikators. Latvijas Dabas fonds, Rīga, 96 lpp.

Klimkowska A., Van Diggelen R., Bakker J. P., Grootjans A. P. 2007. Wet meadow restoration in Western Europe: a quantitative assessment of the effectiveness of several techniques. – Biological Conservation 140: 318–328.

Krause B., Culmsee H., Weche K., Bergmeister E., Leuchner C. 2011. Habitat loss of floodplain meadows in North Germany since the 1950s. – Biodiversity Conservation 20: 2347–2364.

Liepa A., Ķuze J. 2004. Dabas aizsardzībai nozīmīgo Lielupes kreisā krasta posma Sloka–Kalnciems pļavu apsaimniekošanas plāns. Ķemeri, 7 lpp.

Poptcheva K., Schwartze P., Vogel A., Kleinbecker T., Hölzel N. 2009. Changes in wet meadow vegetation after 20 years of different management in a field experiment (North-West Germany). – Agriculture, Ecosystems and Environment 134: 108–114.

Rūsiņa S. 2010. Zālāju biotopi. Grām.: Auniņš A. (red.) Eiropas Savienības nozīmes aizsargājamie biotopi Latvijā. Noteikšanas rokasgrāmata. Latvijas Dabas fonds, Rīga, 131.–181. lpp.

Schaich H., Szabo I., Kaphegyi T. A. M. 2010. Grazing with Galloway cattle for floodplain restoration in the Syr Valley, Luxembourg. – Journal of Nature Conservation 18: 268–277.

Ter Braak C. J. F., Šmilauer P. 2002. CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (version 4.5). Microcomputer Power, Ithaca NY, USA, pp. 500.

Tockner K., Bunn S. E., Gordon C., Naiman R. J., Quinn G. P., Stanford J. A. 2008. Floodplains: critically threatened ecosystems. In: Polunin N. (ed.) Future of Aquatic Ecosystems. Cambridge University Press, Cambridge, pp. 45–61.

The effects of hydrological restoration, mowing and grazing management in Dunduru meadows, Ķemeri National Park

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Summary

The purpose of this paper is to evaluate the effects of the re-meandering of Slampe stream, restoring the floodplain's functions, and grazing and mowing in the restored alluvial grasslands in Ķemeri National Park, Dunduru Meadows. On the basis of monitoring data, changes in grassland vegetation, aquatic communities and bird fauna are analysed and discussed.

Changes in grassland plant communities were caused mainly by grazing, late-summer mowing and paludification. However, eight years after re-establishment of regular management, the grassland is still poor in semi-natural grassland plant species. The increase in plant species richness is limited by land use history in the particular area – amelioration and destruction of the soil seed bank, low species richness in the surrounding grasslands and isolation at landscape level.

Eight years after remeandering, stable aquatic macrophyte communities typical for slow flowing streams have not yet established. Lack of shading in the early successional phases promoted massive development of the macro-algae *Cladophora glomerata*, which successfully occupied the niche of macrophytes. The profile of the stream river bed with relatively steep slopes and without a well-pronounced zone of shallow water has prevented development of littoral and submerged macrophyte communities. Moreover, development of aquatic macrophyte vegetation was affected by the trampling and browsing impacts caused by grazing animals, which in this situation have created more severe damage to aquatic plants than in pastures with stable aquatic ecosystems. The benthic communities in the re-meandered Slampe stream are still poorly structured and do not resemble communities typical for slow flowing natural streams. However, restoration of the floodplain regime has created suitable conditions for migratory birds. A positive trend in Corncrake population was observed.

Introduction

Before intensification of agriculture, large floodplain areas in Europe were covered by fens and alluvial grasslands. During the 20th century, most of them were affected by drainage. Although the human-caused impacts altered wetlands already in the medieval time, the intensity of modifications has dramatically increased during the 20th century (van Diggelen et al. 2006). As a result, 50–90 % of peatlands in Europe were destroyed or severely damaged (Joosten, Clarke 2002; Klimkowska et al. 2007). During the 20th century, many European floodplains were converted into meadows and pastures, i. e., natural ecosystems were transformed into semi-natural, where the associated high species richness has developed and depends on long-term interaction among humans, domestic animals and natural processes.

Modifications in the hydrological regime and establishment of drainage systems have deteriorated most floodplain ecosystems in Europe, including Latvia, also converting them into arable lands and cultivated grasslands. Thus, the seasonal flooding rhythm and natural floodplain functions (e. g. carbon sequestration, regulation of run-off, sediment accumulation, soil development, habitats for wild species) are affected or interrupted (Bischoff 2002; Tockner, Standford 2002; van Diggelen et al. 2006; Urtāne et al. 2012).

Alteration of wetland ecosystems including floodplain grasslands has caused the decline of biodiversity all over Europe (Walker et al. 2004). In comparison to extensive, traditional use of alluvial grasslands for hay collecting or grazing, the massive drainage works during the 20th century, especially in the second half of the 20th century, have damaged many floodplain plant and animal species, communities and their habitats.

Today the semi-natural alluvial grasslands cover only 0.1 % of Latvia (Rūsiņa 2010). During the 20th century, approximately 90 % of them were lost (Sabardina 1958; Rūsiņa 2010). During the Soviet time, particularly in the 1960s and 1970s, vast areas of semi-natural floodplain grasslands were converted into arable lands and cultivated grasslands. This has destroyed not only natural wetlands, but also promoted the decline of wetland-related species, expansion of generalist species (Walker et al. 2004), and decreased landscape diversity.

During the 20th century, numerous small and medium-sized streams were straightened, open ditches and sub-surface drainage systems established, thus interrupting the natural seasonal flooding rhythm and lowering the groundwater table. In 1995, around 1.6 million hectares of land (ca. 25 % of the country) in Latvia were drained (Anon. 1997). Currently, 1500 nationally important streams are recorded in the state register of drainage systems. Their total lengh reaches 21 100 km including 13 100 km of straightened streams and stream stretches (Ministry of Agriculture, unpublished data).

In the 1990s, most drainage systems in Latvia were not well maintained, while new drainage networks were not established due to lack of funding (Anon. 1997). During the last few years, the drainage systems are being gradually renovated. In order to restore degraded wetland ecosystem functions and to re-create habitats suitable for wetland species, since the 1990s, in some protected nature areas, the drainage systems are being blocked (e. g. Bergmanis et al. 2002; Ķuze et al. 2008; Ķuze, Priede 2008).

In Europe, many wetland restoration projects have been carried out (e. g. Bischoff 2002; Donath et al. 2003; Liira et al. 2009). Nevertheless, the results of numerous studies show that the restoration is not always successful in terms of the recovery of the target ecosystem or species (Bischoff 2002; Donath et al. 2003; Klimkowska et al. 2007). It means that restoring a deteriorated wetland is often a challenging task, sometimes it is even impossible.

Re-meandering of the Slampe stream (Ķemeri National Park, Latvia) in 2005 was aimed at increasing the biological capacity and species and habitat richness of the floodplain grassland. It meant restoring the floodplain regime in the Dunduru Meadows, a drained alluvial meadow area, re-creating suitable conditions for the Corncrake *Crex crex*, and maintain the grassland by applying regular management (Ķuze et al. 2008). The project implementation team expected that re-meandering of the straightened stream along with regular grassland management (mowing and grazing) will (1) promote diversification of the grassland plant communities, (2) the floodplain area will become a significant resting place for migrating birds and an important Corncrake breeding area, and (3) the newly created stream will be colonized by communities typical for slow flowing (potamal) natural streams.

The aim of this paper was to assess the success of restoration taking into account different factors that might have affected the results of re-meandering the stream, restoring the floodplain regime and re-establishment of regular grassland management. Several ecosystem components (grassland and aquatic vegetation, birds, macrozoobenthos organisms) were analysed. In this paper we tried to aswer a question: did the restoration actions reach the goal?

Material and methods

Site description

The re-meandered Slampe stream and grassland restoration area is located in the southwestern part of Ķemeri National Park, in Dunduru Meadows (56050'35" N, 23023'58" E), in the floodplains of the Slampe and Skudrupīte streams (Fig. 2). During the 20th century, both the Slampe and Skudrupīte streams were straightened and the flooplains drained and ameliorated.

Although the restoration area and its surroundings are flat, the total stream gradient of Slampe is 34 m, i. e. 1.9 m/km (Zīverts 1998). The stream is slow flowing throughout (potamal). Intensive agricultural lands prevail in the catchment area: the proportion of agricultural lands reaches 50–70 %, 20–30 % of them are intensively used arable lands (Anon. 2009). Therefore, the amount of nutrients leaching into the Slampe stream is high: on average 0.44 t of nitrogen annually per 1 km^2 of catchment area (0.40 t/km^2 from agricultural lands), and on average 0.03 t of phosporus annually per 1 km^2 of catchment area (0.01 t/km^2 from agricultural lands) (Anon. 2009).

Skudrupīte, a small tributary of Slampe stream, and Džūkste stream, which flows downstream from Slampe, were straightened in 1932–1933. Slampe was straightened in the period from 1964 to 1974 (Ķuze et al. 2008). In the 1970s, a sub-surface drainage system was established in Dunduru Meadows. After draining the area, Slampe and Skudrupīte have gained the status of streams of national importance. Therefore, all actions concerning the hydrology of both streams are regulated by the Amelioration Law (the latest edition 01.01.2015.) and the Regulation of the Cabinet of Ministers (23.08.2005., No. 631) „Construction regulation LBN 224-05 „Drainage systems and hydrotechnical constructions””).

In 2005, a 2080 m long stretch of Slampe stream was re-meandered. The action was funded by the European Commission's LIFE-Nature programme, „Conservation of the wetlands in Kemeri National Park, Latvia" project (LIFE2002NAT/LV/8496). The new, re-meandered stream is 4650 m long. Moreover, the water table was elevated by about 1 m and the seasonal flooding regime was restored (Ķuze et al. 2008). After restoration, the area is again regularly flooded during the spring season. Most of the straightened „old" stretch was kept open and at some points connected to the newly created, re-meandered stream.

The width of the re-meandered stream was planned at 3 m (the stream bed, where the water flows throughout the year), the bank slope –1:2, the stream bed decline – 0.1 ‰, and the average depth – 1.2 m (Ķuze et al. 2008). Adjusted calculations of flow capacity show that the re-meandered Slampe stream can carry only the average summer flow rate (Qv. v. = 0.35 m^3/s), while the spring maximum flow rate with exceeding possibility 10 % is about 30 % higher than projected in the technical project of the renaturalization (Grīnfelde, Rituma 2013a). This means that the floodplain can be flooded not only during the spring season and other high water seasons, but also after heavy rain. The floodplain's width (Fig. 1), which was calculated according to the probability of the amount of water and flood frequency, varies from 122.23 (at Qpp 10 % = 11.28 m^3/s) to 145.66 m (at Qpp 1 % = 11.28 m^3/s) at a flooding probability of 10 % (flooding possible once in 10 years) and 1 % (flooding possible once in 100 years). This means that the floodplain grassland can experience not only seasonal, but also permanent flooding (Grīnfelde, Rituma 2013a).

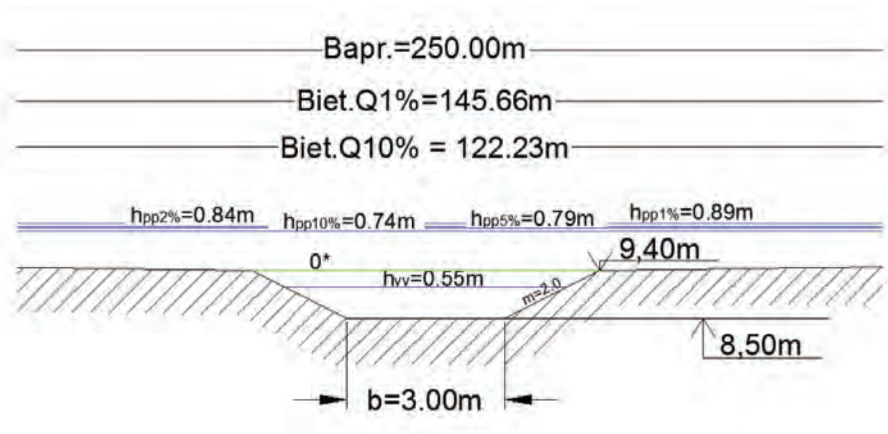

Fig. 1. *The cross profile of the renaturalized stretch of the Slampe stream (source: Grīnfelde, Rituma 2013a). The bankfull width of the renaturalized stretch of the Slampe stream is highlighted with a green line; b – riverbed width, B – calculated bankfull at different flow probabilities, h – water depth at different through flow probabilities.*

When planning the decline of the stream bed and the stream profile, the environmental requirements of the abovementioned Regulation LBN 224-05 (Chapter 9) were not taken into account. The Regulation permits creation of increased stream profile in some stretches to promote formation of sand deposits (Article 368.4). According to regulation LBN 224-05 (Article 268.5), it is allowed to create microstructures (whirlpools, depressions, hiding and spawning places for fish etc.). and the deepest stretches can be differentiated with shallow water zones, also variation in the stream width are acceptable (Article 268.6). This means that in the renaturalization of a state-importance stream the profile can be planned similarly to natural streams if the flow capacity requirements defined in the Regulation are taken into account. In the case of Slampe stream, this was not done. The re-meandered Slampe stream has a regular profile. The stream does not have a shallow water zone, the meander curves were planned without variations in the stream banks. As suggested by the following studies, this was one of the main obstacles limiting successful development of aquatic vegetation.

Not only Slampe and its tributary Skudrupīte, but also other streams in the surroundings have been straightened, the adjacent areas have been drained and converted to agricultural lands. The former floodplain grassland area has been transformed into arable lands and cultivated grasslands. It is surrounded by intensively used, drained agricultural lands on one side and a large forest and bog massif on the other side. Thus, the restoration area is almost isolated from other semi-natural grassland areas. In total the grasslands in the Slampe (Dunduru Meadows) and Skudrupīte floodplains (Melnragu rīkle) cover around 200 ha. In soviet times, both neighbouring floodplain areas were alternately used as grasslands and arable lands, therefore the remains of cultivated grasslands and fallows of different ages may be found in the area.

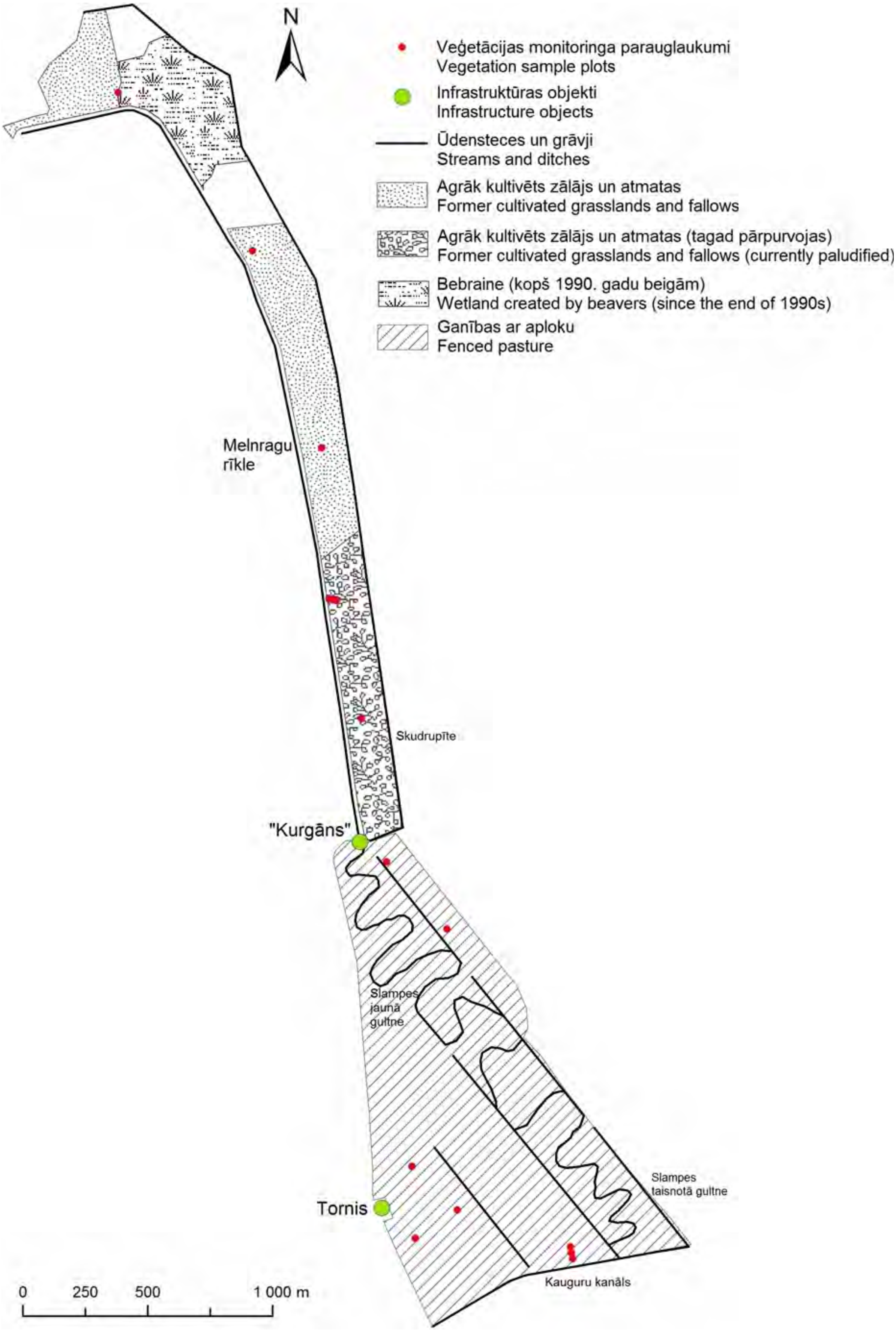

Fig. 2. *Schematic map of the study area: Slampe floodplain (south from "kurgāns") and Skudrupīte floodplain (Melnragu rīkle).*

Grassland management methods

Until the mid 20th century, Dunduru Meadows were used for hay cutting. There are no vegetation records from that time, but the local inhabitants remember the area as covered by „tall, sharp grass", which means that perhaps the area was covered by tall sedge communities. In the 1940–1950s, the grassland was partly overgrown with shrubs, which were later removed and regular management re-established. Since the 1970s, the area has been used as crop fields, hay meadows and a cattle pasture. In the early 1990s, due to restructurization of land ownership and land management the area was largely abandoned. As mentioned in the site management plan of Ķemeri National Park (Anon. 2002), at that time the grassland area was irregularly mowed and used as a cattle pasture, but no regular management of the entire area was applied anymore.

In 2005, a fenced pasture of 110 ha was established and the hardy *Heck* cattle breed and *Konik polski* horses were introduced. Since then, the animals have been grazing in the area throughout the year. The grazing intensity has increased from 0.2 animals per ha in 2005 to approximately 0.75 animals per ha in 2012. During the wintertime the animals are also fed. In the period from 2005 to 2008, the fenced grassland was both grazed and mowed once per year at the end of summer. With the increasing number of animals, the mowing was ceased. The rest of the area (around 60 ha) has only been mowed since 2005, in the first two years the hay was collected, but later on (until 2014) it was chipped and left on the field.

Data collection

Monitoring of flora and vegetation. Flora and vegetation was assessed in Melnragu rīkle (Skudrupīte floodplain) and Dunduru Meadows south from „kurgāns" (Slampe floodplain) (Fig. 1). In 2007 and 2013, a flora survey was done by recording all vascular plant species found. For the species nomenclature, the list of taxa by Gavrilova and Šulcs (1999) was used.

The vegetation monitoring was carried out from 2003 to 2013 in permanent circle-shaped sample plots with a 2 m diameter. All vacular plant species were recorded using the Braun-Blanquet method (Braun-Blanquet 1965) and their cover in percent visually estimated. Until 2008, there were nine sample plots, but this was acknowledged as insuffient, therefore 24 additional plots of 2 × 2 m were established. All plots were surveyed annually in July. Each time all observed changes (e. g. visible hydrological changes, grazing intensity) were recorded.

In order to analyze shifts in vegetation, a total of 241 relevés were used: nine plots were described annually in the period from 2003 to 2013, and 24 plots – annually in the period from 2008 to 2013. All plots were divided into two groups according to the management applied (mowed and grazed since 2005).

In order to analyze the changes in plant species diversity at community level in both management groups (mowed, grazed), a mean number of species per plot was used. Species which characterize certain conditions were selected and divided into three groups: species of wet alluvial grasslands (*Carex acutiformis, C. disticha, C. riparia, C. pseudocyperus, C. vulpina, Galium palustre, G. uliginosum, Lythrum salicaria, Lycimachia vulgaris, Myosotis palustris, Mentha aquatica, Phalaris arundinacea*), ruderal tall perennial herbs (*Arctium tomentosum, Anthriscus sylvestris, Artemisia vulgaris, Cirsium arvense, C. vulgare, Carduus crispus, Urtica dioica*), and low creeping plants and plants typical for heavily trampled soils (*Plantago major, Potentilla reptans, P. anserina, Ranunculus repens, Polygonum aviculare, Trifolium repens*). The sum of their covers was used to illustrate the changes in vegetation.

Biocoenotic structure of aquatic communities. The biocoenotic structure in the re-meandered stream stretch was assessed using two groups of aquatic organisms: aquatic macrophytes and macrozoobenthos. The samples of macrozoobenthos were taken using the standard method LVS ISO 5667-6. According to the type of bottom sediments, the samples were collected using a Petersen grab sampler (size of sample plots 250 cm^2). The samples were fixed with 4 % formaldehyde solution. The macrozoobenthos taxa were identified using a stereomicroscope.

The biocoenotic structure of macrophyte communities was assessed in several 200 m long stream stretches, both in the re-meandered stream and the remnants of the straightened stream. Submerged and free-floating macrophyte samples were taken using a grapnel.

Prior to restoration measures, no detailed assessment of the biocoenotic structure of the stream was carried out (Ķuze et al. 2008). Therefore, the biocoenotic structure in the re-meandered stream was compared to aquatic communities in the remnants of the straightened stream stretch.

Bird counts. In order to assess the suitability of the restored floodplain for migratory waterfowl, in the time period from 2006 to 2013, regular (at least once per week or 10 days) birds counts were carried out annually from March to April. The birds were counted from the artificial hill („kurgāns") in the central part of the area (Fig. 1). Prior to 2006, the bird observations were irregular and non-systematic.

Vocalizing males of Corncrake *Crex crex* were monitored annually since 2003 (once per season in 2003, twice per season since 2004; the methodology is described by Keišs (2006)). Total lengh of the monitoring transect is 9 km covering the northern part of the area (Melnragu rīkle), Dunduru Meadows and Siliņu Meadows (south from Dunduru Meadows). However, in this paper only counts from Dunduru Meadows, the restored Slampe floodplain, which is currently a fenced grazing area (110 ha) were taken into account.

Results and discussion

Shifts in grassland flora and vegetation

Prior to establishment of monitoring, there were no records of vegetation and flora from the area. After ~10 years of the absence of regular grassland management, in 2003 most of the area was described as dominated by species-poor, nitrophilous tall herb communities (Dabas aizsardzības pārvalde, unpublished data). The situation documented in some photos from 2003 suggests that most of the grassland area was covered by a thick layer of dead litter indicating lack of management. Around 2007, most of the area was dominated by plant species typical for cultivated grasslands (mostly *Dactylis glomerata* and *Phleum pratense*) and ruderal tall perennial herbs of classes *Artimisietea vulgaris* and *Galio-Urticetea* (e. g. *Anthriscus sylvestris, Urtica dioica, Arctium tomentosum, Artemisia vulgaris, Calamagrostis epigeios, Elytrigia repens*) (Ķuze et al. 2008; A. Priede, unpublished data). Very few indicators of biologically valuable semi-natural grasslands (Anon. 2013) could be found except in small patches, which most probably were never cultivated or cultivated several decades ago and since then were managed as grasslands by mowing or grazing.

In 2007, the total number of vacular plant species in the area was estimated as 154, in 2013 – 199 taxa (except woody species and aquatic macrophytes). However, these numbers are not high considering the size of the area. In 2013, 14 indicators of biologically valuable semi-natural grasslands (Anon. 2013) were found in a ca. 200 ha area: *Agrimonia eupatoria, Carex panicea, Dactylorhiza baltica, Galium boreale, Geranium palustre, Filipendula vulgaris, Ophioglossum vulgatum, Pimpinella saxifraga, Plantago media, Polygala amarella, Primula veris, Ranunculus auricomus, Sesleria caerulea* and *Trollius europaeus*. Most of these species were found in small numbers and were present only in a tiny proportion of the area, perhaps being remnants from the pre-drainage period. Several indicator species have established within a small-scale grassland restoration experiment by sowing seeds of indicator species (Priede 2012), because they could not be found outside the experimental plots.

Insignificant changes in the vascular plant flora were observed during the last eight year period, though the number of species has increased. Major changes were observed in the vegetation structure, not in the species composition.

In the mowed part of the area (Skudrupīte floodplain) ruderal tall perennial herbs and a few graminoids typical for cultivated grasslands prevailed (e. g. *Dactylis glomerata, Phleum pratense*). Around 2013, the ruderalized vegetation was largely replaced by a mosaic of tall sedges (*Carex acutiformis, C. disticha, C. riparia, C. vulpina*) and *Phalaris arundinacea*. However, the ruderal tall perennial herb communities (*Galio-Urticetea, Artemisietea vulgaris*) with a high proportion of a few graminoids were present. In the last years, an elevated water table in Skudrupīte floodplain was observed, which has caused paludification and subsequent shifts in vegetation toward the *Bidention tripartitae* community, a pioneer community typical for enriched still and sluggish waters and damp disturbed sites. In some patches, the graminoids *Elytriga repens* and *Agrostis gigantea* formed nearly mono-dominant stands. Development of *Bidention tripartitae* community was promoted not by seasonal inundation, but by a constantly elevated water table since 2010, when a large part of the meadow was flooded from the end of summer until next spring. As a result, in the following two years the former grassland and tall herb vegetation was replaced by a mosaic of open patches without any vegetation and sparsely vegetated *Bidention tripartitae* patches. Later on, the damp depressions gradually overgrew with tall sedges, while the elevations were gradually occupied by the former ruderalized grassland vegetation, though a little richer in species of alluvial grasslands (Fig. 3).

Remarkable shifts in vegetation occurred in the pasture of the Slampe floodplain. The current tendency leads toward a diversification of species composition. The mean number of species per sample plot has gradually increased since 2007 (three years after the introduction of grazing animals) (Fig. 4). The change in the number of species is more or less biased as the monitoring was carried out by three different persons (2003 to 2005 by the first monitoring performer, 2006 – by the second, and since 2007 – by the third). However, the data from the last years suggest that the vegetation is becoming richer in species and the vegetation structure more diverse. The mean number of vascular plant species in the mowed part (Skudrupīte floodplain) was highly variable during the monitoring period without a tendency to increase (Fig. 4).

Probably the different tendencies in the mowed and grazed part are caused not so much by different management methods (grazing vs. mowing without hay collection) as by changes in the hydrological regime. In the last years, the mowed part is also affected by standing water during the vegetation

2008: the vegetation is dominated by a few graminoids (mostly Agrostis gigantea and Elytrigia repens).

2010: the area was partly flooded at the end of summer. Water standing in the meadow for a long time including the vegetation season caused rapid changes in vegetation.

2011: due to standing of water, the vegetation became patchy with unvegetated patches, Bidention tripartitae pioneer communities and gradual development of tall sedge stands.

2013: paludification promoted establishment of graminoids and tall sedges, which are typical in wet alluvial grasslands and fens.

Fig. 3. *Turnover of plant communities in the mowed area due to hydrological changes (standing water). View from the same location in different years. Photos: A. Priede, A. Liepa.*

season causing paludification, which became visible in the vegetation since 2010, although the vegetational changes are perhaps also a late response to grass chipping, which is often blamed for causing a decline of species diversity within a few years after its regular application.

A tendency of increasing species richness was also observed in the „new" sample plots (since 2008) in the pasture. In the mowed part of the grassland, due to the paludification effects caused by standing water since 2010, the mean number of species per plot was fluctuating without a certain tendency (Fig. 4). In the pasture, the vegetation shifts could be interpreted as diversification, though slow, while in the mowed grassland area, particularly in the topographical depressions, the vegetation is still unstable.

As a result of the former land use and following abandonment, both in the mowed and grazed part of the grassland, in the period from 2003 to 2008 there was a high proportion of ruderal tall perennial herbs and fallow plant species. Since 2006, both in the mowed and grazed area, the proportion of ruderals has notably decreased. In the mowed part, the ruderal tall perennial herbs are being replaced by tall sedges and graminoids, in the pasture – mostly by graminoids, legumes and low creeping, trampling-resistant plants (Fig. 5).

Some patches, particularly in the mowed part, are still dominated by ruderal tall perennial herbs, which can most probably be explained by the effects of late mowing and grass chipping in the last years. Late mowing (in August or the beginning of September) allows ripening and spreading of ruderal tall herb seeds, particularly of *Anthriscus sylvestris*.

Decline of the mean species number and formation of hydrological conditions unfavourable for development

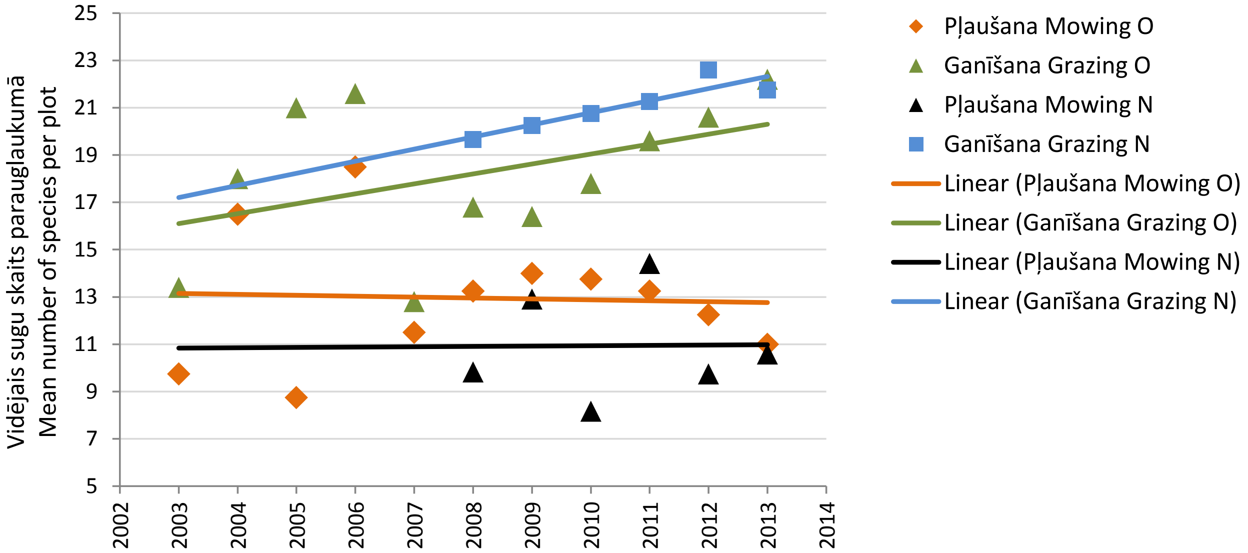

Fig. 4. *Changes in the mean number of plant species per sample plot in mowed and grazed areas (O – plots monitored since 2003; N – plots monitored since 2008; the yellow vertical line depicts the introduction time of pasture animals, the red line – beginning of regular mowing, and the dark blue line – increased effect of standing water in the mowed part of grassland).*

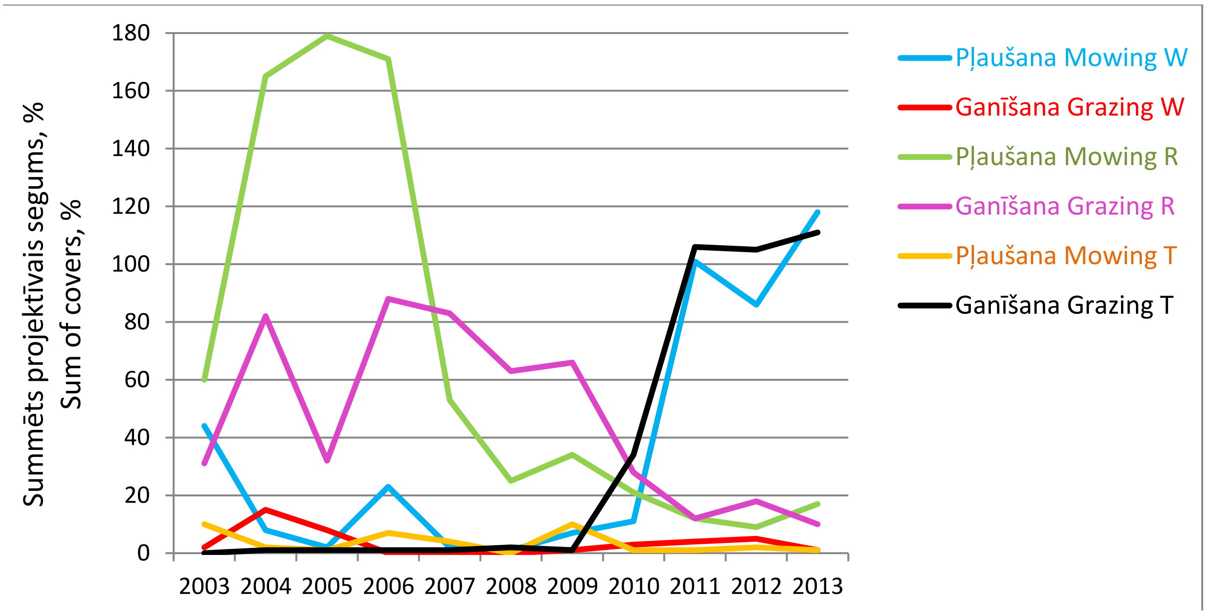

Fig. 5. *Changes in the sum of covers of wet floodplain grassland (W), ruderal tall herb (R) and low creeping, trampling-resistant plant communities (T), 2003– 013.*

of semi-natural grassland vegetation (paludification) in the mowed part of area suggest that the management has not achieved the expected results yet – recovery of semi-natural grassland. It means that late mowing is an ineffective management method in ruderalized alluvial, hydrologically altered grasslands. It does not help to diminish the dominance of tall ruderal perennial herbs, to increase the richness of plant species and to establish typical species of alluvial semi-natural grasslands.

Compared to late mowing with chipping, grazing management has achieved better results when using plant species richness, proportion of ruderal tall perennial herbs and the establishment of low creeping plants as indicators (Fig. 3., Fig. 4). For example the dominance of Anthriscus sylvestris has significantly decreased since 2006, mostly as a result of grazing in the pasture (Fig. 6) and elevation of the water table in the mowed part. Since 2007, the mean cover of this species has decreased from › 30 % to ‹ 5 %.

Beginning of June, 2006.

Beginning of June, 2013.

Fig. 6. *Decline of Anthriscus sylvestris in the grazed area, view from the artificial hill in 2006 and 2013 during the blossoming season of the species. Photos: A. Liepa, A. Priede.*

Significant differences in vegetation succession after re-establishment of management were caused by grazing intensity. Although the animals are able to access all the fenced area, the spatial patterns of grazing and trampling differ seasonally and diurnally. It is reflected in the vegetation composition and structure. The most intensive grazing pressure was achieved in the southern part of the grassland area near Kauguru kanāls stream, where the animals stay for most of day during the summer. Therefore, the sward there is short and heavily trampled. Some other parts of the area are visited rarely during the summer, thus the proportion of ruderal tall herbs is still high, though being more and more grazed in late summer, which might cause a vegetation shift toward pasture plant communities.

In summary, the major drivers causing shifts in vegetation in the Dunduru Meadows is re-establishment of management (predominantly grazing) and changes in the hydrological regime (paludification) (Fig. 7).

Assessment of management effects on grassland vegetation

Establishment of a drainage system, and following cultivation of the floodplain from the 1960s to the 1990s, significantly altered the floodplain ecosystem and its functions, the soil properties and plant communities destroying the typical species complex. During the last 11 years, the vegetation in Dunduru Meadows was affected by different management

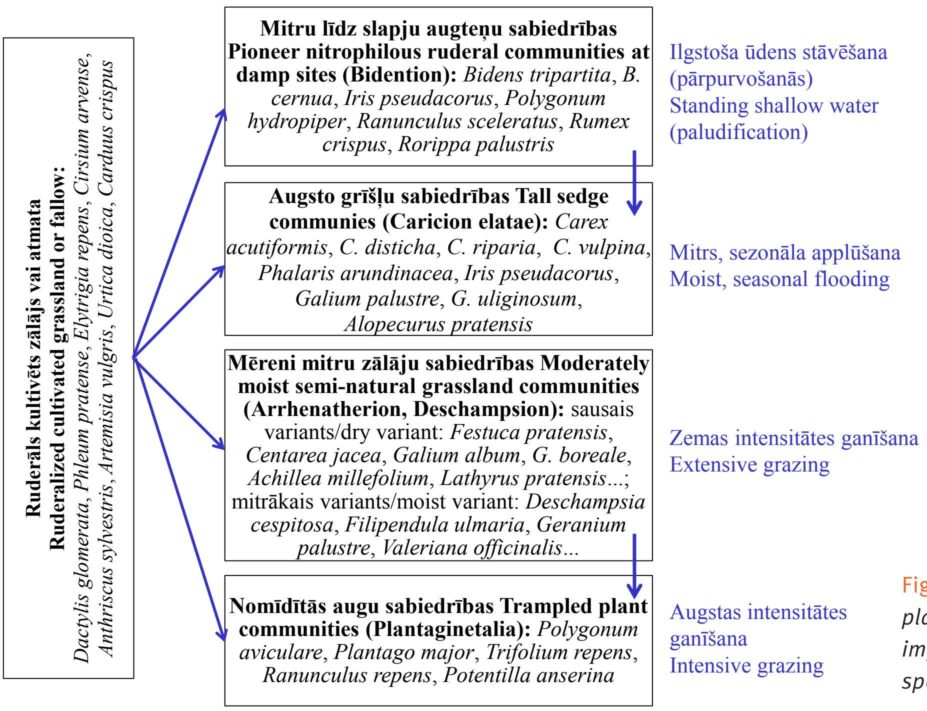

Fig. 7. *Major shifts in grassland plant communities under different impacts. Only the most frequent species are shown.*

methods and lack of management during the first 2–3 years of the monitoring period. The vegetation of the alluvial grassland is recovering, though the process is slow and the plant communities still lack many typical species of alluvial grasslands. Even relatively common species of alluvial grasslands are absent from the area or they are still very rare. However, some tendencies, e. g. decline of ruderal tall herbs indicate improvement of the grassland habitat. The ruderal tall herbs are being gradually replaced by lower sward graminoids, sedges and other plants of semi-natural grasslands. The proportion of low creeping, trampling-resistant plant cover and a decrease of the mean vegetation height in the pasture indicates the impact of grazing and trampling. The grazing intensity differs among different sections of the pasture, both spatially and seasonally. The areas adjacent to the streams and near the winter feeding grounds are more intensively trampled and grazed, which promotes development of uneven mosaic-type vegetation patterns. The growing number of animals promotes an increasing expansion of low sward trampled plant communities. At the grazing intensity of 2013, most probably this type of vegetation will soon dominate in the area.

Restored seasonal flooding in Slampe floodplain in the fenced grazing area seems to have a minor impact on vegetation shifts in comparison to grazing. However, re-meandering of Slampe stream has caused changes upstream in Skudrupīte floodplain (Grīnfelde, Rituma 2013b). This has resulted in flooding, not only during the spring season, but also an increased water table during the summers, consequently affecting the vegetation.

The soil disturbances caused by digging of the new stream bed in 2005 created only a short-term impact in a relatively narrow belt along the re-meandered stream. The disturbed soil was shortly dominated by annual ruderal plant communities, but within 1–2 years they were replaced by perennial vegetation similar to the surrounding grassland.

Late mowing and chipping is often criticized as an unsuitable method for management of semi-natural grasslands (Gaisler et al. 2004; Rūsiņa 2010), though some authors suggest that it is probably applicable and economically justified if combined with hay collecting in some years (e. g. Liira et al. 2009). Our experience in Dunduru Meadows cannot justify either the first or second opinion on the basis of scientific evidence due to the monitoring design – the monitoring was carried out in a real management situation without control plots. The management was mixed and not constant during the monitoring period. Moreover, the mowing and chipping impacts cannot be seperated from the hydrological change, which affected all mowed areas during the monitoring period.

Comparing late mowing with chipping and grazing, grazing seems to be a more effective method in the restoration of degraded alluvial grasslands. Almost 10 years of mowing did not achieve good results. Nevertheless, it is important to take into account why late mowing was chosen as a management method in this particular case. This method was selected because the area is important for the Corncrake population, which could be negatively affected by early mowing, and the manager applied for agri-environmental payments, which required late mowing. Hay collection in late summer was not done due to its low nutritional value and limited possibilities to dry and collect the hay at the end of summer. In order to mow annually, agri-environmental payment requirements were compulsory, as there was no alternative funding for management, but in this case it was not the best solution for grassland biodiversity.

One of the significant obstacles in successful restoration of plant diversity in Dunduru Meadows is the geographical location of the area, its historical use in intensive agriculture and the surrounding intensive, highly fragmented agricultural landscape. As suggested by some studies (e. g. Wagner et al. 2003), the soil seed bank is sufficient for restoring a species-rich grassland only in areas where the traditional management has been recently ceased, whereas in intensive agricultural lands the restoration potential is significantly lower. In Dunduru Meadows, the restoration measures have brought notable improvements, however, the methods applied (restoring the hydrological regime, mowing, grazing) were insufficient in achieving the ideal result – a species-rich semi-natural alluvial grassland. During the cultivation period, the soil seed bank has become poor. In this area, due to limited migration possibilities (surrounding landscape patterns, lack of natural streams, hay transport etc.) the recovery of plant species diversity is limited. In some European countries, this kind of limitation is being compensated with additional measures like spreading a species-rich hay, transplanting species-rich sods or sowing seeds collected in species-rich semi-natural grasslands (e. g. Klimkowska et al. 2007; Rūsiņa 2010). Similar measures would also be useful in Dunduru Meadows. In Skudrupīte floodplain, seeds from species-rich grasslands were applied in some experimental plots in 2008 (Priede 2012). However, the scale of this experiment was too small to achieve a diversification of species over larger areas.

Riparian communities

Aquatic macrophytes. Approximately eight years after re-meandering, the communities of aquatic macrophytes were still not stabilized and typical for slow flowing streams. On the stream banks, the vegetation is composed of plant species of alluvial grasslands. In the re-meandered stream, the shallow water zones are covered by sparse vegetation formed by plants of damp sites and small patches of submerged macrophytes. It is not typical vegetation for slow flowing streams, because it is still under development and characterizes early successional phases. Only in some stretches of the re-meandered stream *Mentha aquatica, Myosotis palustris* and *Phragmites australis* are present on the banks. In the submerged and free-floating vegetation belt, only *Glyceria fluitans, Lemna trisulca* and *L. minor* are present.

In comparison to the re-meandered stream, vegetation in the straightened „old" stretches of Slampe stream, which currently function as oxbows, is richer in species: *Phragmites australis, Phalaris arundinacea, Sium latifolium, Scirpus lacustris, Lycopus europaeus, Veronica anagallis-aquatica*, and several *Carex species* are present here. In the deeper water zone *Myriophyllum spicatum, Lemna trisulca, L. minor* and *Elodea canadensis* are common.

In the spring flooding period, the re-meandered stream is connected to the straightened stretch, which was not filled up during the restoration. The aquatic vegetation in the straightened stretch is comparatively rich. Thus, a poor seed bank might not be the reason for the limited success of re-vegetation in the re-meandered stream. Slow re-vegetation is defined by a combination of several factors. In the early successional phases, due to a slow exchange of water and a lack of shading, the stream warmed up during the summer season resulting in a massive overgrowth of macroscopic green algae *Cladophora glomerata* (Ķuze et al. 2008), which in turn hindered development of macrophytes. The steep banks and the profile of the stream bed without a shallow water zone prevented development of submerged macrophyte communities (Fig. 8). However, these two factors could be the only ones limiting development of macrophyte communities. This is indicated by the vegetation in the straightened stretches of Slampe stream, which has a trapeze-shaped profile, and the banks are also not shaded by shrubs. Also, the lack of a shallow water zone and full light conditions did not prevent development of emergent vegetation. Thus, we assume that the presence of grazing animals in the area is a significant factor limiting the development of macrophyte communities in the re-meandered stream. Most probably, trampling and browsing cause more severe impacts on macrophytes in the early successional stages than on stable macrophyte and riverbank plant communities.

In order to promote development of vegetation on the newly created stream banks, willows were planted in approximately a 100 m long stretch in April, 2009 (Dabas aizsardzības pārvalde, unpublished data). But, since willow plantings were not fenced, this action was unsuccessful and most of the plantings in the pasture did not survive and do not create shading as expected.

Macrozoobenthos. Lack of macrophyte communities characteristic for potamal streams has also defined the structure and composition of macrozoobenthos communities. The studies of communities in renaturalized streams suggest that the community might stabilize within 300–5000 days (Williams 1977; Urtāns 1989; Urtāne 1992; Nielsen 1995; Ripl et al. 1995). Eight years after re-meandering, the structure of macrozoobenthos communities in Slampe stream is still simplified and not typical for potamal streams.

The depth of the re-meandered stream is on average 0.5–0.7 m. The stream bed is sandy and muddy, with a small proportion of detritus. The biomass of macrozoobenthos is relatively high – 82.25 g/m^2. A majority of organisms are molluscs Mollusca (71.6 m/m^2) followed by Sialis spp. (3.0 g/m^2) and Asellus aquaticus (2.5 g/m^2). Other groups of organisms form a very small proportion with a tiny biomass (Fig. 9).

Simplified structure and low species diversity was basically defined by the lack of macrophytes and trampling effects. The composition of inverteberate species in the re-meandered stream is more characteristic for standing waters and ditches: *Cloeon dipterum, Caenis horaria, Asellus aquaticus, Sialis* spp., *Tabanus* spp. No caddisfly *Trichopera*, dragonfly *Odonata* and beetle *Coleoptera* individuals were found in the samples of the re-meandered stream. Typical species of slow flowing streams – oligochaetes *Oligochaeta* and chironomids *Chironomidae* were found in very small numbers. Oligochaetes were represented only by bottom-dwelling species, while no phytophilous chironomids were found in the re-meandered stream. Although the nutrient load from the catchment is high (Anon. 2009), there is still no natural mud layer on the bottom, therefore the number of oligochaetes is low.

Although the total biomass of macrozoobenthos organisms in the straightened stream was lower in comparison to the re-meandered stream (26.9 g/m2), all species groups typical for potamal streams were found in the samples. The dominant group of organisms was molluscs (110 g/m2) followed by dragonflies and crustaceans, and the dominant species – Sphaerium corneum (Fig. 9).

Floodplain grassland as a habitat for migratory birds

After restoration of the Slampe floodplain in Dunduru Meadows, spring floods were observed in 2006, 2009, 2010, 2011 and 2013. In the first season after restoration in 2006, the number of birds counted was comparively low – 33 ducks belonging to four species (*Anas*

The agglomerations of Lemna minor and Spirodela polyrrhiza within a renaturalised stretch of the Slampe stream characterizes a simplified and unstable macrophyte community.

An emergent aquatic plant zone has not developed yet. On river banks, patches of Glyceria fluitans are characteristic for early successional stages.

An emergent plant zone along the former regulated stretch of the Slampe stream is formed by dense Phragmites australis stands. Stream bed profile is trapeze-shaped with steep side slopes and water depth > 1 m.

Stands of submerged water plants in the straightened stretch of the Slampe stream is formed by Myriophyllum spicatum and Lemna trisulca. Above them, Lemna minor and Spirodela polyrrhiza forms separate patches.

Fig. 8. *The structure of macrophyte communities in the re-meandered stretch of the Slampe stream and in the straightened stretch of the Slampe stream. Photos: L. Urtāne.*

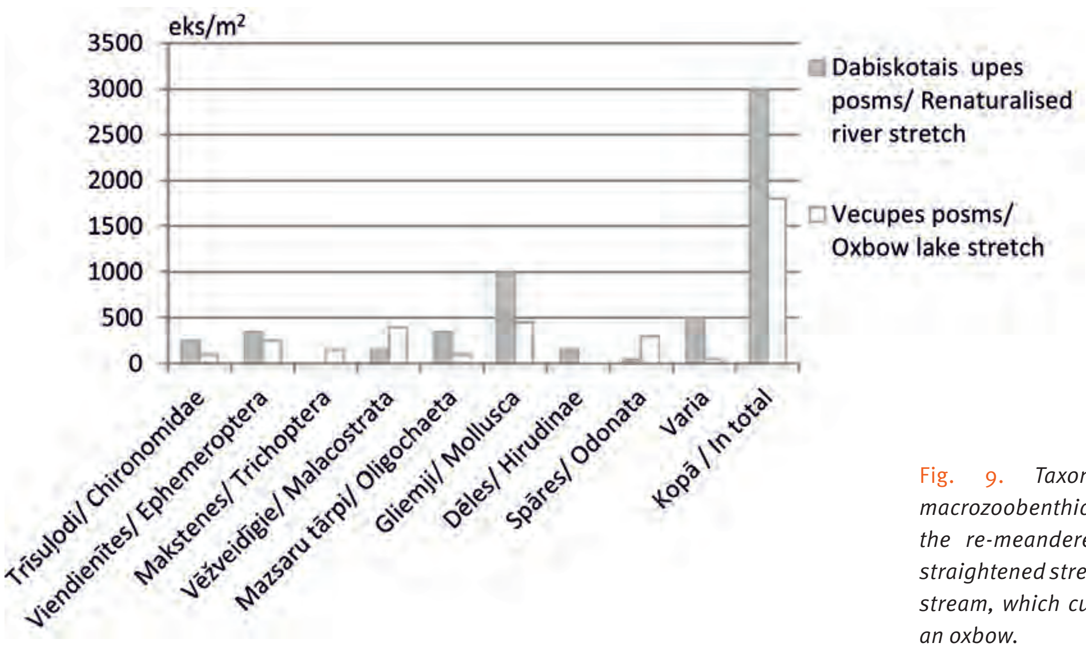

Fig. 9. *Taxonomic group of macrozoobenthic organisms in the re-meandered and previously straightened stretches of the Slampe stream, which currently function as an oxbow.*

platyrhynchos, A. crecca, A. acuta and *Bucephala clangula*) (Ķuze et al. 2008). In the following two years no significant floods and concentration of waterfowl was observed. In 2009–2011, the area experienced high inundation during the spring, overflooding most of the Dunduru Meadows. A large number of migratory waterfowl was observed in 2009, when on the 1st of April it reached approximately 500 individuals (*Anas platyrhynchos* (410 ind.), *A. penelope* (50 ind.), *A. acuta* (20 ind.), *A. crecca* (20 ind.) and one pair of *Anas clypeata*. Similarly, a large number of ducks was recorded on the 31st of March, 2010 (90 % of the total number of all individuals were represented by *Bucephala clangula, Anas penelope, A. crecca* and *A. acuta*).

The largest number of birds in Slampe floodplain grassland was recorded on the 5th of April, 2011: approximately 1000 ducks (*Anas platyrhynchos, A. crecca, A. clypeata, A. acuta*), 300 ind. of *Vanellus vanellus*, 16 ind. of *Cygnus cygnus*, 4 ind. of *C. columbianus*, 4 ind. of *Grus grus*, 2 ind. of *Branta canadensis*, 1 ind. of *Ardea cinerea*, and 1 ind. of *Ciconia ciconia*. The ducks stayed over a large floodplain area – both in the northern part of the the Slampe floodplain grazing area and in the southern part of the Skudrupīte floodplain grassland. Two days later, on April 7th, the number of birds was estimated by R. Matrozis, M. Kalniņš and A. Poppels (R. Matrozis, unpublished data) as 2500 ind. of *Anas platyrhynchos*, 1400 ind. of *A. crecca*, 400 *A. acuta*, 100 *A. penelope*, 2 ind. of *A. strepera*, 1 ind. of *A. querquedula*, 2 pairs of *A. clypeata* and 2 ind. of *Bucephala clangula*. Additionally, ca. 700 ind. of *Vanellus vanellus* were counted, which means that the total number of birds during this count exceeded 5000.

In 2012, the floods were not well-pronounced, and a gathering of ducks was not observed. However, on the 26th of March, ca. 2000 geese were counted: *Anser albifrons* (prevailed) and *A. fabalis* (in smaller numbers). This was the first case, at least since establishment of the Ķemeri National Park, that such a large number of geese was observed. Gathering of geese in the Dunduru Meadows and the surrounding areas was also recorded in autumn, 2013 (4th of October), when during the low water season 4000–5000 ind. were counted. However, the majority of them stayed outside the floodplain on the harvested croplands to the south of Kauguri canal and to the northwest of the canal bridge, while only about 500 geese stayed in the grazing area in the Slampe floodplain.

Comparatively large floods occurred in 2013, however, the spring was late and snowy, thus the flood season began late, around mid-April. Most probably this explains the low number of birds recorded in 2013: on the 14th of April only 150 ind. of *Anas platyrhynchos,* 2 pairs of *A. acuta*, 3 ind. of *Ardea cinerea* and 1 ind. of *Phalacrocorax carbo* were counted.

Prior to floodplain restoration, a concentration of migratory birds was not observed in Dunduru Meadows. Therefore, in our opinion, the gathering of the large numbers of birds (up to 5000 ind. at one time) can be interpreted as a good result. Restoration of the Slampe floodplain has created an excellent resting place for migratory birds. In terms of importance, the area at the bird maximums can be compared to the downstream lowland of Svēte River or some shallow coastal lakes. In Dunduru Meadows, the maximum number of birds can be observed only for a short time – usually a few days, which is related to the flooding regime. The number of days when the Slampe area is flooded is shorter than in, for example, the Lielupe floodplain near the mouth of the Svēte River, which is defined by the size of the river and the density of the drainage network in the cathment. During the years with less pronounced floods, the number of migratory birds counted at once was low (up to a few hundred), predominantly *Anas platyrhynchos* and *A. crecca*, which usually stay near the banks of the re-meandered stream or in the reed stands along the old drainage ditches.

The floodplain as a breeding area for the Corncrake

One of the aims in restoring the Slampe floodplain grassland area was the creation and maintenance of suitable habitats for the Corncrake. Before implementation of the restoration project, it was expected that during the re-meandering work the number of Corncrakes might temporarily decline as they might be affected by the disturbances caused by earthwork, but in the long-term the number of Corncrakes will increase as the habitat properties improve.

No direct impact on the Corncrake population caused by earthwork (digging of the new meanders in the beginning of 2005) was found. Already in the first season after re-meandering, the number of counted birds was larger than the year before. Perhaps this can be explained by the relatively small areas directly disturbed by the earthwork. The mean number of Corncrakes in 2005–2013 was 7.4 males/km^2, maximum number – 11.28 males/km^2 (in 2008). The number of males counted in 2012–2013 was on average 9.77 males/km^2. The maximum number of males recorded in Latvia of 18.75 pairs/km^2 (Keišs 1997a) was not reached due to grazing impacts in Dunduru Meadows. The sward height suitable for the Corncrake is only available in some parts of the fenced area. At the same time, the Corncrake density recorded in Dunduru Meadows is remarkably higher than the average in semi-natural pastures in Latvia – 1.46 pairs/km^2 (Keišs 1997b).

Conclusions and recommendations

Designing the stream bed

In the renaturalization of a stream, designing the new stream bed and calculation of the flow parameters, one should take into account not only the drain rate, but also the peculiarities of stream functioning and the ecological requirements of species directly affected by the hydrological regime in the floodplain.

During the planning stage, the habitat experts should define certain conditions suitable for the target species and communities concerning the hydrological regime. In order to create conditions suitable for restoring alluvial grasslands and Corncrake habitats, the maximum flow rate (with different exceeding probabilities), mean summer flow rate and maximum summer-autumn flood flow rate should be such that the area is flooded only during the high water season in spring. In this case, restoration of the Slampe floodplain has caused an elevated water table upstream in the Skudrupīte floodplain causing paludification of the meadow.

In order to ensure functioning of the renaturalized stream, the stream bed profile should be similar to natural stream bed profiles taking into account the flow requirements. The stream bank slopes should be designed in a way that there is a shallow water zone. The meanders should be planned similar to natural conditions, taking into account the velocity profiles: a deeper water zone in the outer part of the meander loops and a shallow water zone in the incurved meander loops. This would promote development of aquatic macrophyte vegetation and would ensure faster stabilization of aquatic communities.

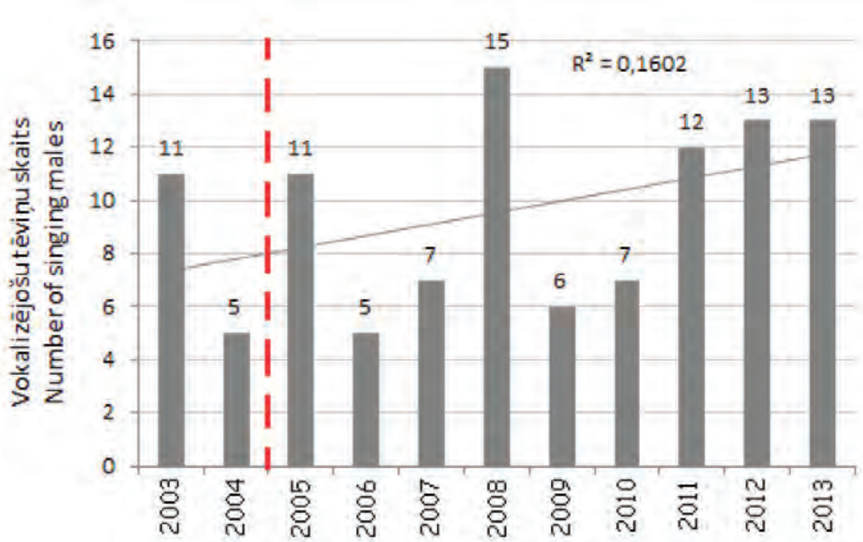

Fig. 10. *Number of Corncrakes in Dunduru Meadows (2003–2013). The dotted line shows the time of floodplain restoration.*

Impact of hydrological restoration to other related elements of the hydrological network

In cases when only a certain stretch of a stream is being restored, one must bear in mind that restoration measures will most probably cause changes upstream or downstream and might alter other elements related to the hydrological network. In Slampe floodplain, restoration of the hydrological regime in the floodplain grassland area brought changes upstream in the floodplain of the tributary Skudrupīte. As a result, the water table in the upstream meadows increased. However, in this case it might also be a temporary effect, which will be averted during the ongoing project aimed at restoring the hydrological regime of Skudrupīte stream and floodplain.

Necessary management measures

Eight years after implementation of the restoration project and re-meandering the stream, there was still no stable aquatic vegetation typical for slow flowing streams. Its development was hindered not only by the stream bed profile described above, but also by the lack of shading in the early successional phases and the trampling impact caused by grazing animals. In order to ensure development of aquatic plant communities, it is advisable to create shading by planting shrubs along the stream banks (Urtāns, Urtāne 2011). The trampling effect could be diminished by fencing some stretches of the re-meandered stream, which would also help to protect the shrub plantings from animal browsing. Fencing some parts of the grassland and stream might also improve the conditions for one of the target species, the Corncrake.

Reaching all targets at once is impossible

The example of Dunduru Meadows shows that benefits for all target ecosystem components are difficult or even impossible to reach, especially concerning vegetation and birds. In Dunduru Meadows, grazing has created a positive effect on vegetation structure and composition. At the same time, the grazing intensity is too high for grassland birds, especially the Corncrake. During an eight year period, late mowing seems to be ineffective in restoring plant diversity in a deteriorated floodplain grassland, the proportion of ruderal species is still high. At the same time, it is a suitable method for managing Corncrake habitats.

Similar conclusions derive from observations in the Skudrupīte floodplain, where the restoration of the Slampe stretch has caused a raised water table. Currently it is a suitable nesting habitat for waders and an important resting habitat for ducks during migration. In the spring of 2012, 4–5 pairs of *Vanellus vanellus* were breeding in this part of the area, and during migration numerous individuals of *Tringa glareola* and up to 300 ducks *Anas* spp. (predominantly *Anas platyrhynchos*, *A. penelope* and *A. crecca*) were recorded. Though being an important area for birds, the high water table and ongoing paludification process does not allow development of alluvial grassland vegetation, which develops only under seasonal flooding conditions in interaction between natural process and moderate human-caused impacts. On permanently water-logged soils the temporary pioneer communities will be most probably replaced by tall sedge fen communities, similar to those of beaver ponds and adjacent wetlands. Though this wetland might be highly important for many wetland species including threatened bird species, this kind of area cannot be called and managed as a semi-natural grassland, at least not using tradicional management methods.

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References

Anon. 1997. Irrigation in the countries of the former Soviet Union in figures. Food and Agricultural Organization of the United Nations, Rome.

Anon. 2002. Ķemeru nacionālā parka dabas aizsardzības plāns. CarlBro, Rīga.

Anon. 2009. Lielupes baseina apgabala apsaimniekošanas plāns. Latvijas vides, ģeoloģijas un meteoroloģijas centrs, Rīga.

Bergmanis U., Brehm K., Matthes J. 2002. Dabiskā hidroloģiskā režīma atjaunošana augstajos un pārejas purvos. Grām.: Opermanis O. (red.) Aktuāli savvaļas sugu un biotopu apsaimniekošanas piemēri Latvijā. Vides aizsardzības un reģionālās attīstības ministrija, Rīga, 49.–56. lpp.

Bischoff A. 2002. Dispersal and establishment of floodplain grassland species as limiting factors in restoration. – Biological Conservation 104 (1): 25–33.

Braun-Blanquet J. 1965. Plant sociology: The study of plant communities. Hafner, London.

Diggelen, van R., Middelton B., Bakker J., Grootjans A., Wassen N. 2006. Fens and floodplains of the temperate zone: present status, threats, conservation and restoration. – Applied Vegetation Science 9: 157–162.

Donath T. W., Holzel N., Otte A. 2003. The impact of site conditions and seed dispersal on restoration success in alluvial meadows. – Applied Vegetation Science 6 (1): 13–22.

Gaisler J., Hejcman M., Pavlů V. 2004. Effect of different mulching and cutting regimes on the vegetation of upland meadow. – Plant, Soil and Environment 50 (7): 324–331.

Gavrilova Ģ., Šulcs V. 1999. Latvijas vaskulāro augu flora. Taksonu saraksts. Latvijas Akadēmiskā bibliotēka, Rīga, 136 lpp.

Grīnfelde I., Rituma A. 2013a. Slampes upes hidroloģiskā režīma izmaiņas pēc renaturalizācijas darbu veikšanas. LIFE10 NAT/LV/160. Ķemeru nacionālā parka hidroloģiskā režīma atjaunošana. Projekta atskaite.

Grīnfelde I., Rituma A. 2013b. Skudrupītes hidroloģiskais un hidrauliskais režīms. LIFE10 NAT/LV/160. Ķemeru nacionālā parka hidroloģiskā režīma atjaunošana. Projekta atskaite.

Joosten H., Clarke D. 2002. The wise use of mires and peatlands. International Mire Conservation Group and International Peat Society.

Keišs O. 1997a. Griežu uzskaišu rezultāti Latvijā 1989.–1995. gadā. – Putni dabā 7 (1): 11–21.

Keišs O. 1997b. Griežu *Crex crex* skaita novērtējums un biotopu izvēle Latvijā 1996. gadā. – Putni dabā 7.1: 22–27.

Keišs O. 2006. Naktsputnu uzskaites lauksaimniecības zemēs. LU Bioloģijas fakultātes Zooloģijas un dzīvnieku ekoloģijas katedra, Latvijas Ornitoloģijas biedrība, Latvijas Dabas fonds, Rīga. Pieejams: http://lob.lv/lv/programmas/naktsputni.php

Klimkowska A., Van Diggelen R., Bakker J. P., Grootjans A. P. 2007. Wet meadow restoration in Western Europe: a quantitative assessment of the effectiveness of several techniques. – Biological Conservation 140: 318–328.

Ķuze J., Liepa A., Urtāne L., Zēns Z. 2008. Palienes režīma atjaunošana Slampes upes lejtecē. – Auniņš, A. (ed.) Aktuālā savvaļas sugu un biotopu apsaimniekošanas problemātika Latvijā. Latvijas Dabas fonds, Rīga, 41. –51. lpp.

Ķuze J., Priede A. 2008. Raising of water table in areas influenced by drainage in Ķemeru Mire, Latvia: methods and first results. In: Pakalne M. (ed.) Mire conservation and management in especially protected nature areas in Latvia, pp. 106–115.

LVS ISO 5667-6. 2005. Ūdens kvalitāte. Paraugu ņemšana. 6. daļa: Norādījumi paraugu ņemšanai upēs un ezeros. Latvijas Nacionālā standartizācijas institūcija „Latvijas standarts".

Liira J., Issak M., Jõgar Ü., Mändoja M., Zobel M. 2009. Restoration management of a floodplain meadow and its cost-effectiveness – the results of a 6-year experiment. – Annali Botanici Fennici 46: 397–408.

Nielsen M. B. 1995. Restoration of streams and their riparian zones – South Jutland, Denmark. In: Eiseltova M., Biggs J. (eds), Restoration of stream ecosystems: an integrated catchment approach. IWRB Publication, pp. 30–44.

Priede A. 2012. Dabiska zālāja atjaunošanas eksperiments. Ģeogrāfija mainīgajā pasaulē. IV Latvijas ģeogrāfijas kongress. Latvijas Universitāte, Rīga, 179–181.

Ripl W., Hildmann C., Janssen T., Gerlach I., Heller S., Ridgill S. 1995. Sustainable development of a river and its catchment. The Stor River, Germany. Restoration of stream ecosystems: an integrated catchment approach. IWRB Publication 37. 76–111.

Rūsiņa S. 2010. Dabisko zālāju apsaimniekošana augāja daudzveidībai. Grām.: Auniņš A. (red.) Aktuālā savvaļas sugu un biotopu apsaimniekošanas problemātika Latvijā. Latvijas Dabas fonds, Rīga, 29.–43. lpp.

Sabardina G. 1958. Latvijas PSR dabisko zālāju klasifikācija. LPSR ZA izdevniecība, Rīga, 38 lpp.

Tockner K., Stanford J. A. 2002. Riverine flood plains: present state and future trends. – Environmental Conservation 29 (3): 308–330.

Urtāne L. 1992. Biocenožu attīstības īpatnības pēc Jaunupes rekultivācijas. Vides aizsardzība Latvijā, 117. –127. lpp.

Urtāne L., Urtāns A., Bērziņš E. 2012. Ūdensceļi un ūdensmalas. Vadlīnijas ūdenstilpju un ūdensteču izmantošanas un apsaimniekošanas plānošanai. Vidzemes plānošanas reģions, 210 lpp.

Urtāns A. 1989. Mazo upju atjaunošanas perspektīva. – Zinātne un Tehnika 8.

Urtāns A., Urtāne L. 2011. Praktiski padomi, kā uzlabot ūdensteču funkcionalitāti. Global Water Partnership Central and Eastern Europe – GWP CEE, 30 lpp.

Wagner M., Poschlod P., Setchfield R. P. 2003. Soil seed bank in managed and abandoned semi-natural meadows in Sooma National Park, Estonia. – Annali Botanici Fennici 40: 87–100.

Walker K. J., Stevens P. A., Stevens D. P., Mountford J. O., Manchester S. J., Pywell R. F. 2004. The restoration and re-creation of species-rich lowland grassland on land formerly managed for intensive agriculture in the UK. – Biological Conservation 119: 1–18.

Wiliams D. D. 1977. Movements of benthos during the decolonization of temporary streams. – Oikos. Bd. 29.

Zīverts A. 1998. Slampe. Enciklopēdija „Latvijas Daba" 5, 116. lpp.

Experience with stream restoration in the Netherlands as an example for European lowland streams

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Introduction

The vast majority of streams in the Netherlands can be considered as lowland streams. Lowland streams are low-energy rivers with a channel slope smaller than 1 m/km, a channel width smaller than 20 m, and a channel bed consisting of sand or peat (van der Molen et al. 2012).

Estimations are that only about 20 % of the lowland streams in the Netherlands have a natural origin. Most lowland streams in the country were developed in the 18th and 19th century, mainly for the reclamation of swamps and forests for agricultural use. In other areas waterways were created to mine groundwater for watermills and laundry works. In this period, nutrient-rich stream water was also used to fertilize agricultural fields. Farmers constructed hydraulic structures, with a purpose to stimulate inundation.

In the mid-20th century, intensifying agricultural activities led to introduction of industrialized fertilizers, groundwater management and channelization of the majority of lowland streams. Streams were redesigned to become straight in the context of scaling. Meanwhile cross-sectional areas were increased to tackle increased flows and weirs were built to control water levels, especially during dry seasons.

In the past decades it became clear that channelization had destructive consequences for the aquatic and terrestrial ecology of lowland streams. The destruction resulted from decreased currents, a loss of diverse habitats and bad water quality (lack of oxygen and high nutrient levels in particular). This led to the start of a period of restoration. Stream restoration in the Netherlands has been accelerated by the Water Framework Directive (WFD) and its objective to achieve a good ecological state. Another important objective for stream restoration, which follows from the National Water Act (Waterbeleid voor de 21ste eeuw: WB21), is to increase the retention of water within the catchment to reduce flood risks (as a result of climate change).

Common practice in the Netherlands

Stream restoration in the Netherlands often involves construction of sinuous channels (re-meandering), imitating the channel planform characteristics before channelization (Eekhout 2014). The sinuous planform is often based on historical sources (e.g. detailed maps from the 18th and 19th century). Re-meandering or other channel reconfiguration measures are often applied at a local scale, in channel reaches.

The design procedures for restoration adopted in the Netherlands follow a process-based approach (Eekhout 2014). The new channel dimensions aim to satisfy several requirements simultaneously: reducing flood risk in downstream areas, maintaining the existing ground water levels for adjacent agricultural fields and improving the aquatic ecology. Most commonly, a 1D-flow model is used in the design of the cross-sectional shape of the new channel, focussing on all three main requirements of the new channel at the same time. Often, this turns out to be too complex, with ecological requirements (partly) given up.

In most stream restoration projects lowered floodplain areas are constructed surrounding the streams (Figure 1). Lowered floodplains typically also boost natural terrestrial floodplain processes.

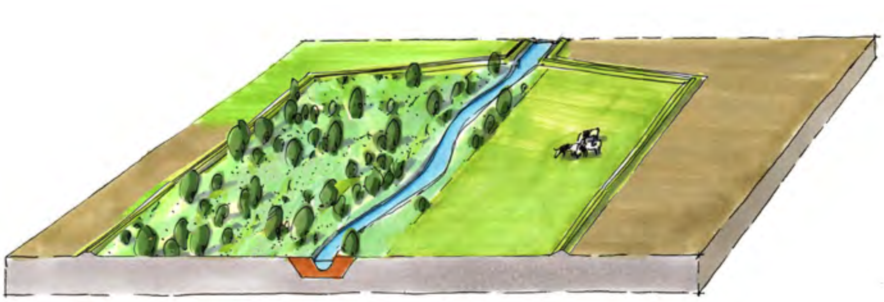

Figure 1. *Lowered floodplain (agricultural or natural) for water retention (drawing: Dirk Oomen, Stroming BV).*

Basic approaches to stream restoration

While restoring streams over the last 30 years, a number of basic approaches were applied in the Netherlands.

With patience, by letting the river do the work

Depending on the longitudinal profile, discharge and type of soil of the riverbed, a canalized stream will in time start re-meandering by itself. Obstacles in the streambed – like beaver dams and burrows, fallen trees and curves – stimulate this process. To allow this natural process to take place, artificial riverbank protection has to be removed. Burrow activities of mammals like beavers weaken the riverbank and increase erosion. In a meandering riverbed, places with fast flowing water alternate with slow flowing areas. Vegetation growing in the slow flowing areas forces the stream to curve even more and raises the water level and as a result the buffer capacity. Fallen trees play an important role in this process; to speed up meandering, trees can be placed into the riverbed.

A tree fallen into the stream forces the flow through a narrow gap between the uplifted roots and the remaining riverbank, causing quick erosion. Worm, October 2003 (photo: Willem Overmars, Stroming BV).

By restoring the old riverbed

Redirecting the canalized stream into old meanders (when still present) is another way of restoration. This is realized by blocking the artificial stream; redirected flowing water will search its own path, erosion and sedimentation will soon give a natural result. To start these processes, it might be necessary to remove fresh sediment or vegetation overgrowing the historical riverbed, or even dig out the curves of the original river course. The removed material can be used to fill and block the canalized streambed and the draining ditches in the floodplain. To maximize ground water level rise, the artificial channel should be blocked on both ends.

In 2010 the old meander 'Mölnmarsch' was reconnected with the river Vecht (photo: Waterboard Vechtstromen).

Digging a new river bed

If the old streambed is no longer visible in the landscape, or if it has turned into an oxbow lake with certain ecological values, the best solution might be to dig a completely new meandering riverbed. The floodplain topography will guide the design, leading the new river through natural depressions in the floodplain.

Stream restoration measures only mark the beginning of the restoration process. Natural processes like erosion and sedimentation will modify the initial situation to a natural equilibrium. In order to 'let nature do the work' enough space will be needed. How much, depends on the local situation. In natural areas this doesn't cause problems; in agricultural areas however, adjacent areas will have to be protected to minimize negative effects, e.g. by creating lowered floodplains, see Figure 1. In most cases maintenance is needed in the initial phase of development.

Lowland rivers: swamp or stream?

Main focus point for the restoration of lowland streams is the delicate balance between active bed-forming processes on the one hand and sedimentation and vegetation on the other. Larger slope and discharge (stream power) in combination with fine and loose sediment will enable self-sustaining river systems (at the right side of balance in Figure 2). A low stream power in combination with availability of light and nutrients will result in vegetation growth and sedimentation of fine sediments at the left hand. Without maintenance activities this will result in the forming of marsh and swamp.

Restoration measures, taken in a certain river-stretch of stream, need to comply with the physical situation. When the balance tends to bend to the right, restoration measures need to focus on the right dimensions of the bed (corresponding a bank full discharge occurring once in 1.5–2 years), as well as to the prevention of vegetation growth by reducing nutrient levels and/or creating adjacent tree zones in order to reduce the amount of light falling onto the water. When the balance tends to the left the stream power is too small to prevent vegetation overgrowing the bed and the situation will tend to a swamp. In order to enable vegetation growth and keep maintenance activities on a minimum in this swampy situation, the dimensions of the bed should be larger and shallower.

New river bed guided by low areas in the floodplain. Peizerdiep, 2014 (photo: Bart Reeze, Stroming BV).

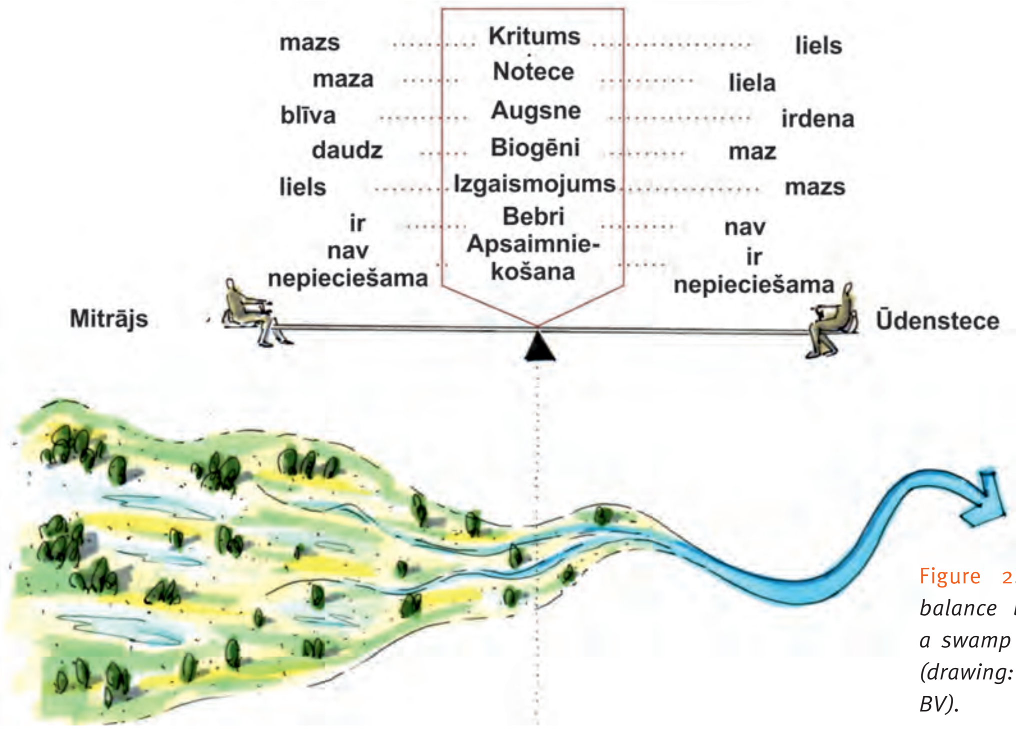

Figure 2. *Aspects affecting the balance between development of a swamp (left) or a stream (right) (drawing: Dirk Oomen, Stroming BV).*

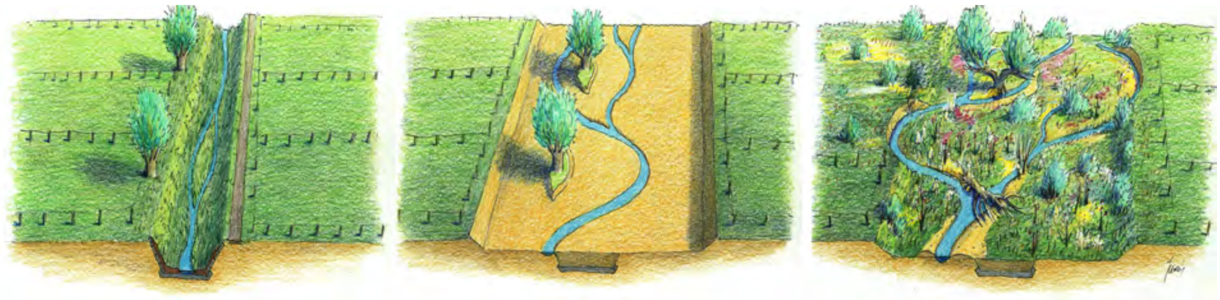

Figure 3, *The Roode Beek in Limburg, an example of a new excavated bed in which the stream could find its own course. Left: former situation, middle: just after the excavation, and right: after 10 years of natural processes (drawing: Jeroen Helmer, ARK Natuurontwikkeling).*

Examples and results

In the Netherlands there is a lot of experience with restoration of rivers and smaller streams. Since the 1990's, especially in the eastern and southern part of the country the water boards have restored natural courses of hundreds of smaller streams. In most cases the old bed was no longer visible and new bed had to be created. An example of this type of restoration is the Roode Beek near Schinveld in the province of Limburg (Figure 3). To activate natural processes a broad shallow bed was excavated with enough room for the stream to shape its own path.

Aerial view of the Roode Beek in 2012, 10 years after the restoration (photo: Dirk Oomen, Stroming BV).

In the relatively high parts of the Netherlands there are examples of streams where digging was not necessary and only the artificial riverbank protection had to be removed. Due to the steeper slope (2–3 m/km, which is steep for the Netherlands) relatively strong currents occur and the stream will more easily erode its banks and form new meanders. In this case digging is not necessary, as the stream has enough strength to restore itself (Figure 4).

Another example of stream restoration without engineering can be found in the Lage Raam. In this lowland stream (slope < 1 m/km) maintenance stopped 15 years ago letting 'nature do the work'. Since then the bank vegetation slowly narrowed the stream bed (see below). This led to problems in 2014 when a flood event urged the regional water authorities to eliminate some of the vegetation. Unfortunately, no clear improvements of the ecological status were observed during these 15 years, probably because of the regulated discharge regime and high nutrient levels.

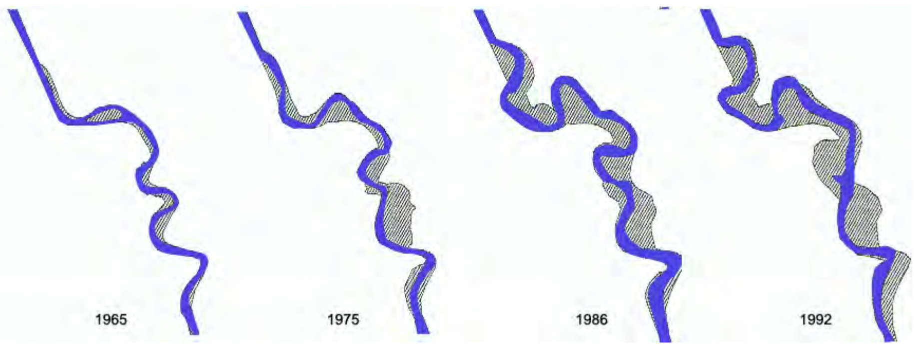

Figure 4. *History of the meandering of the Geul in Limburg (blue is the streambed, grey the eroded part of the valley floor). The Geul is an example of a small river with a relatively steep slope. After the maintenance of the banks stopped (in 1970), natural processes became active again and the river made several new meanders, thus enlarging its valley (drawing: Jeroen Helmer, ARK-Natuurontwikkeling).*

More recently, tests were done with (tree) stems in a number of streams in the Netherlands, e.g. Hierdense Beek, Tongelreep and Jufferbeek. A protocol was set up to optimise ecological effects and to keep backwater effects limited. Per location a couple of trees crossing each other were lowered into the stream, covering the whole width and 10–20 meters length of the stream. The results are promising: an increase of habitat and stream fauna diversity was observed while backwater effects were limited. However, in agricultural areas without tree cover and higher nutrient levels the location might become overgrown by aquatic macrophytes and bank vegetation.

Developing bank vegetation (Glyceria maxima) after stopping maintenance. Lage Raam, 2013. At the bottom right the same stream at a section with normal maintenance (photos: Dirk Oomen, Stroming BV).

Recent developments

5 Zones-concept

As a response to some of the major pitfalls for river restoration, Verdonschot (2009) introduced a new concept for river restoration at the catchment level. Verdonschot distinguishes five zones in a stream valley: 1) the stream itself, 2) the adjacent tree zone, 3) the shrubbery zone as a transition zone between the tree zone and the buffer zone, 4) the buffer zone with cultivated grassy vegetation and 5) the upper part of the stream valley consisting of cultivated and urbanised areas (Figure 6). Restoring the stream at a catchment level means strategic planning of the different zones in the landscape. The concept might be helpful in recognising the existence and distribution of the actual zones in the catchment area and measures that may need to be taken (e.g. the development of missing zones).

Introduction of stems in the Hierdense beek (photo: Bart Reeze, Stroming BV).

Building with Nature

Building with Nature is a term introduced by a consortium of contractors and consulting companies and scientists, combining the development of new

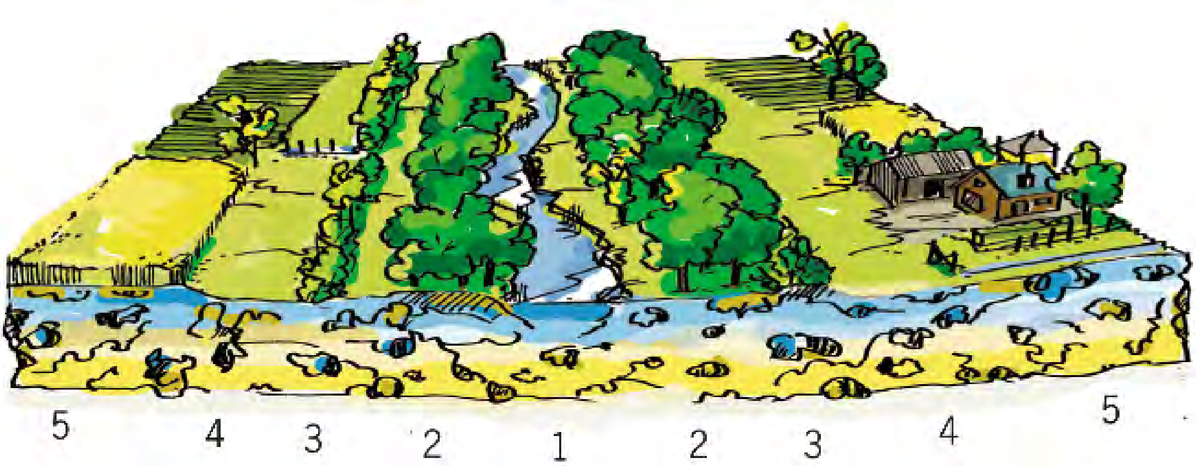

Figure 6. *Graphic representation of the five zones in a stream valley, according to Verdonschot (2009).*

concepts with practical realisation power. The general idea behind Building with Nature is to utilize natural processes and provide opportunities for nature while realising hydraulic infrastructure. Examples are the development of marshland outside dikes to improve the safety levels of those dikes and the creation of a 'sand engine' to help protect parts of the Dutch coast, while encouraging the development of new dunes and the flora and fauna associated with them (www.ecoshape.nl).

With respect to streams, the Building with Nature concept very much resembles the natural restoration approaches mentioned in this article, with its strong focus on utilizing natural processes. An example of such a Building with Nature projects is the river bed elevation in the Hierdense beek by adding sand humps at some locations along the stream letting the stream pick up and distribute the sand during peak discharges.

River bed elevation by slowly pushing sand into to the stream, letting the 'water do the work'. Hierdense beek, 2014 (photo: Bart Reeze, Stroming BV).

Lowland streams show little morphological activity

In his study on three traditional stream restoration projects (Hagmolenbeek, Lunterse beek and Tungelroyse beek) and a man-made canal (Gelderns-Nierskanaal) Eekhout (2014) demonstrated that morphodynamics occurred mainly in the first year after construction. After initial morphological adjustments, the channel planform remained stable. Eekhout concludes it is likely that the sinuous planform observed on historical maps is a result from exogenous influences, rather than autogenous processes. In general, the studied lowland streams showed little morphological activity. The rapid establishment towards an equilibrium state of the channel planform is at odds with the view on lowland streams as small rivers migrating actively in their own deposits. Thus, the term re-meandering may be misleading, because it appears that the morphological activity in restored stream mostly is too small to renew the process of meandering.

Key-factors for river restoration success

Finally we conclude with five key-factors for river restoration success deduced from Didderen et al. (2009). In this study a number of stream restoration projects in the Netherlands were evaluated. The key-factors identified are:

1) Focus on the catchment scale. Until now, stream restoration in Europe has mainly focussed on local measures (Eekhout 2014). Common practice in the Netherlands is no exception to this. However, river restoration success is more likely when the whole catchment (stream valley) is taken into account.

2) Take all abiotic environmental factors, and their mutual dependencies, into account when improving stream conditions Knowledge of the dynamic linkages between forms and processes across different scales in natural streams and rivers is the key to understanding in-stream heterogeneity, and such understanding is essential to restore streams/rivers to natural conditions (Pedersen et al. 2014).

3) Be clear about the restoration goals. SMART objectives must be set early on in the project (SMART: Specific (concrete, detailed, well defined), Measureable (quantity, comparison), Achievable (feasible, actionable), Realistic (considering resources) and Time-Bound (a defined time line)). Firstly, determine the overall aim of the project, then define how you are going to do this or measure success. SMART integrated objectives reflect ecological and hydro-morphological measures and links (Hammond et al. 2011). Restoration goals need to be coherent and in line with the environmental context.

4) Involve stakeholders and the local community. Every river restoration has got its technical challenges, but the social component is just as important. A stream corridor restoration plan should reflect and combine stakeholder interests and establish a framework to facilitate communication among all involved and interested parties. Ensuring the involvement of all partners and beginning to secure their commitment to the project is a central aspect of "getting organized" and undertaking a restoration initiative (FISRWG 1998).

5) Discover the 'genius of the place'. Although there are many general principles, every stream is different and has its own specific qualities. There is no common river restoration plan that can be applied to all river systems: general principles need to be adapted to the local circumstances.

References

Didderen K., Knegtel B., Verdonschot P., Besse-Lototskaya A. 2009. Enquête beek(dal)herstelprojecten 2004–2008. Evaluatie van beekherstel over de periode 1960–2008 en analyse van effecten van 9 voorbeeldprojecten. Directie Kennis en Innovatie, Ministerie van Landbouw, Natuur en Voedselkwaliteit, Ede. Rapport DKI nr. 2009/dk125-O

Eekhout J.P.C. 2014. Morphological Processes in Lowland Streams. Implications for Stream Restoration. Wageningen University, Wageningen. PhD thesis. 178 p.

FISRWG 1998. Stream Corridor Restoration: Principles, Processes and Practices. Federal Interagency Stream Restoration Working Group. GPO item no. 0120-A. Sudoc's No. A 57.6/2:EN3/PT 653.

Hammond D., Mant J., Holloway J., Elbourne N., Janes M. 2011. Practical River Restoration Appraisal Guidance for Monitoring Options (PRAGMO). Guidance document on suitable monitoring for river and floodplain restoration projects. River Restoration Centre, Cranfield, Bedfordshire.

van der Molen D.T., Pot R., Evers C.H.M., van Nieuwerburgh L.L.J. [red]. 2012. Referenties en maatlatten voor natuurlijke watertypen voor de Kaderrichtlijn Water 2015–2021. Eindversie. Rapportnummers STOWA 2012-31.

Pedersen M.L., Kristensen K.K., Friberg N. 2014. Re-Meandering of Lowland Streams: Will Disobeying the Laws of Geomorphology Have Ecological Consequences? PLoS ONE 9(9): e108558. doi:10.1371/journal.pone.0108558

Verdonschot P.F.M. 2009. Het brede beekdal als klimaatbestendige buffer in de veranderende leefomgeving: flexibele toepassing van het 5B-concept in Peel en Maasvallei. Alterra Wageningen UR, Wageningen.