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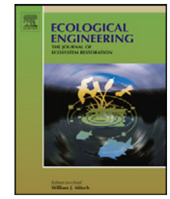

A GIS-based model for testing effects of restoration measures in wetlands: A case study in the Kampinos National Park, Poland

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ABSTRACT

Valuable wetland areas in the Kampinos National Park (KNP), Poland, are exposed to anthropogenic pressure mainly from the neighbouring Warsaw metropolitan area. Deterioration of hydrogenic soils and wetland vegetation has been observed for many years in the KNP. Currently land use change (urbanisation) and increasing groundwater intake are regarded as the main threats to the KNP wetland ecosystems. The objective of this study is developing an integrated GIS-based tool, that allows cause–effect analysis and prediction for the endangered wetland ecosystem in the KNP. The analyses were carried out for two wetland belts occupying the area of 177 km^2. The results indicate that soil and vegetation types vary with the groundwater depth and the level of degradation of habitats corresponds to lowering groundwater levels. The approach allows predicting change in the degree of degradation of soils and vegetation in terms of probabilities conditioned on groundwater depth. The results indicated that only one of five considered restoration scenarios, the one assuming exclusion of the whole drainage network in the study area, could significantly improve wetland state.

1. Introduction

Natural areas, such as forest and wetlands surrounding cities or densely populated regions are in multifunctional use and tourism and outdoor recreation is often an important recognised function (Bell et al., 2009). The Kampinos National Park (KNP) is located close to Warsaw, the capital of Poland. The main objective of this national park is protection of wetlands with their rich flora and fauna, however, urban pressure poses a threat to meet this objective. Over 2000 people live in the Park area and own much land there (Markowski et al., 2011), and it is visited by nearly 50,000 tourists per year (Słomka et al., 2006) mostly from the Warsaw metropolitan area. These three functions, nature protection on one hand and settlement and tourism on the other hand, are obviously in conflict. Cater (1995) suggested that the connection between tourism development, socio-economic development and the environment is circular and cumulative. Breaking the negative links between tourism and the environment is very difficult and needs constant trade-offs. Juutinen et al. (2011) concluded from their choice experiment application conducted in the Oulanka National Park in Finland that increasing tourism facilities, if combined with shrinking biodiversity, are welfare reducing managerial actions in national parks.

During the past two centuries, wet areas in the park have been drained and turned into agricultural lands, which reduced significantly natural wetland habitats. The drainage system is an important factor behind decreasing groundwater levels in the KNP (Krogulec, 2003a; Mioduszewski et al., 2010). The water level in the River Vistula has also decreased, which contributed to unfavourable lowering of the groundwater table (Mioduszewski et al., 2010). Flood protection dikes along the River Vistula, constructed in the last century, reduced water replenishment in the area to precipitation only, enhancing dry conditions (Gutry-Korycka, 2003; Gutry-Korycka et al., 2011). All this, together with individual resident houses construction in the park and its buffer zone that augmented during the last decades, lead to degradation of these valuable natural areas representing the "city lungs" for Warsaw.

Given the importance of the KNP for preservation of wetland flora and fauna, as well as its recreation functions for Warsaw metropolitan area, there have been many investigations aimed at preventing further degradation of the wetlands and their restoration (Chmielewski, 1997; Mioduszewski et al., 2010). However, most of the results of such investigations are qualitative (e.g. maps, inventories, etc.) and no attempt was made to integrate the existing findings in a tool that allows both testing different hypotheses about the main factors behind degradation and possible effects of

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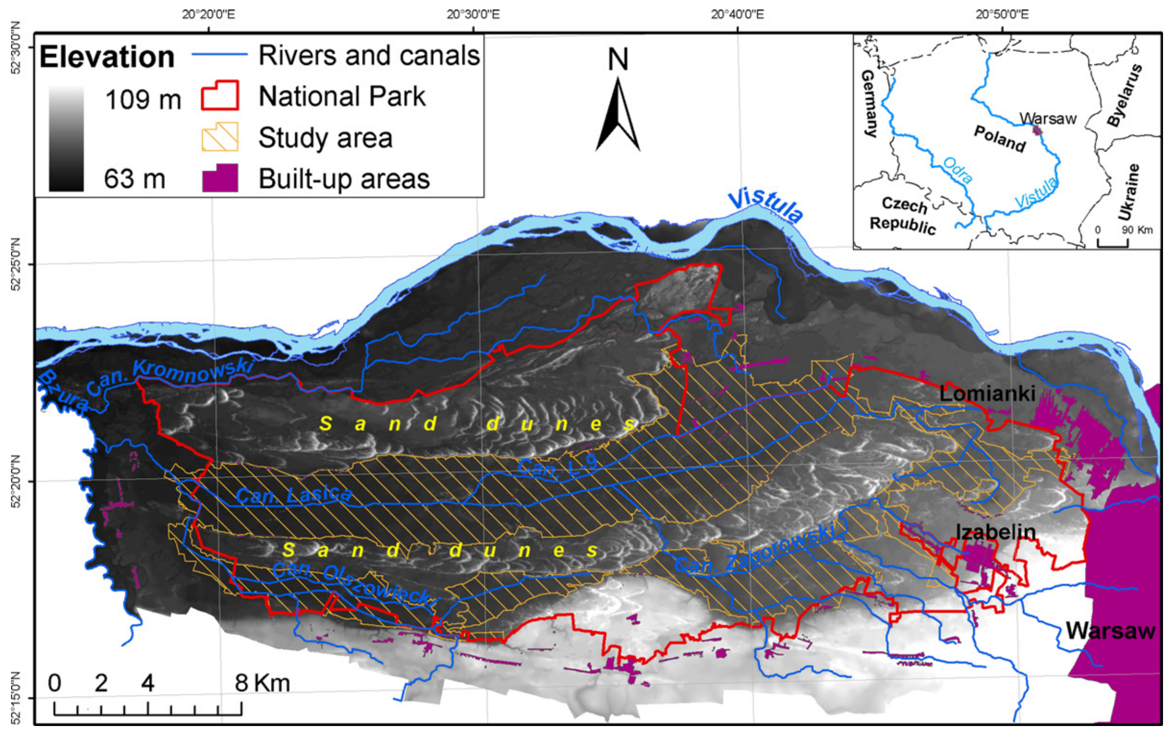

Fig. 1. Location of the Kampinos National Park and the study area.

amelioration. The objective of this study is to assess this gap creating a GIS-based model useful for testing the effect of restoration measures, first of all raising groundwater table, in wetland national parks basing on available qualitative and quantitative observations. GIS-based management tools are common practice in national parks (Chormański and Wassen, 2005; Mahesh et al., 2007; Kardel et al., 2009; Chormański et al., 2009), including KPN, where a hydrogeological spatial database GIS-KPN (ArcView 3.x platform) was developed (Rossa, 2003). Whiteman et al. (2010) used GIS-based approach for risk assessment of groundwater-dependent terrestrial ecosystems in England and Wales. Alexandridis et al. (2007) used GIS techniques for selection of the best restoration scenario for a lake-wetland ecosystem degraded due to human activities.

Two basic topics will be addressed in this study, namely: basic hypotheses about the functioning of the wetlands in the park and the possible causes of their deterioration, and impact of specific measures for restoration and preservation of the wetlands.

The suggested model is based on a system perspective of the wetlands. Dooge (1967) gives the following definition of a system: "A system is any structure, device, scheme, or procedure, real or abstract, that interrelates in a given time reference, an input, cause, or stimulus, of matter, energy, or information, and an output, effect, or response, of information, energy, or matter". Here we will mainly address the cause–effect perspective. A fundamental task is to identify and quantify those variables that can explain the present situation of the wetland system, i.e. to find cause–effect relations. This implies analysing basic hypotheses about the functioning of the wetlands in the KNP, i.e. that the spatio-temporal variation of vegetation cover and soil types of the wetland system is mainly caused by climate and hydrological conditions but can also be human induced. Precipitation observations may serve as a fundamental indicator of the natural variability of climate conditions, as it is not influenced by local factors. Also groundwater level fluctuations follow mainly the natural variability in rainfall input to the wetland system but can also be affected by local man-induced changes, both direct, like drainage and changed land use, and indirect, like pressure from tourism. The ambition is to verify this hypothesis by means of joint analyses of precipitation and groundwater data (cause variables) and data on soils and vegetation cover (effect variables) in order to test the response of these latter to restoration measures.

2. Geographical setting

Kampinos National Park is situated in central Poland on the Middle Mazovian Lowland in the pre-valley of the River Vistula. It is bordered by the River Bzura to the west, by the River Vistula to the north and by the city of Warsaw and its suburbs to the east and south (Fig. 1). The KNP was founded in 1959 for protection of the unique mosaic of landscape consisting of dunes, wetlands and forests. It covers a total area of 385.44 km^2. In 1977 the buffer zone area of 377.56 km^2 was established around the National Park. The KNP with its buffer zone is recognised by UNESCO as Kampinos Forest Biosphere Reserve and is one of the numerous NATURA 2000 areas in Poland.

The whole area of the park consists of interchanging belts of dunes and marshes oriented east-west and parallel to the River Vistula. The park includes 90% of the Canal Łasica catchment, which is the right tributary of the River Bzura. The system of canals and ditches drains both southern and northern marsh belt areas. Predominant ecosystems in KNP are forests (about 70% of the area), which are mostly coniferous or mixed. They cover the dune area and rarely the marsh area. The marsh area is covered by a mosaic of non-forest habitats including surface and submerged open water communities—reedbeds, sedgelands, fens, transitional mires, bogs, moist and fresh meadows, pastures, *Nardus* grasslands, heathlands (Michalska-Hejduk, 2001; Andrzejewska, 2003). The study is focused on wetland areas of the KNP (marshes) occupying ca. 177 km^2, for which detailed spatio-temporal data described in Section 3 were available. This is the part of the Kampinos National Park where groundwater is predominantly shallow, hence it has direct influence on the soil moisture and wetland state. Fig. 1 shows the

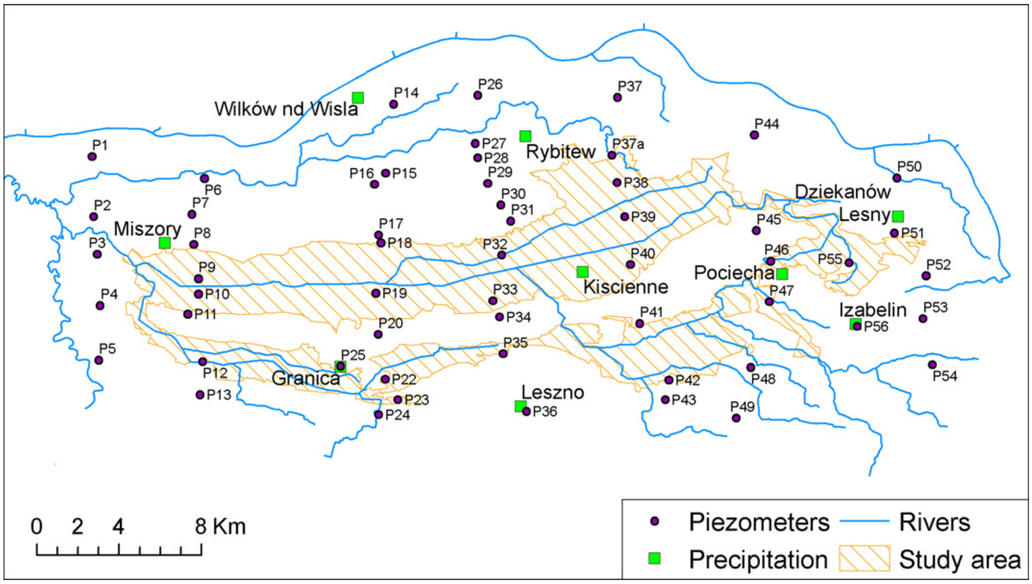

Fig. 2. Location of piezometers and precipitation stations used (locations can be viewed also at interactive web GIS portal http://levis-map.sggw.pl/kpn/).

location of the study area and the KNP, main surrounding built-up areas, the Digital Elevation Model (DEM) and the hydrographical network.

3. Materials and methods

3.1. Data

Data used in the study are of two different types: time series over precipitation and groundwater levels and maps based on inventories of soils and vegetation in KNP. These data were jointly analysed in the frame of DEM.

3.1.1. DEM

The DEM of the KNP and its buffer zone (Fig. 1) was generated using the Topo to Raster interpolation tool available in ArcGIS 9.x, based on the approach of Hutchinson (1996). The data source for the model generation was a set of elevation contour lines (primary data source) and elevation points elaborated in the digital form based on topographic map in scale of 1:10,000. Interpolation parameters have been adapted to the scale of the data source with post-interpolation enforcement on. In order to create a dataset representing as much as possible the reality and for improving the interpolation results the DEM was burned by the layer of river network. The DEM was created with a spatial resolution of 20 m, and then resampled to 100 m suitable for application in the model.

3.1.2. Climatic and groundwater data

Daily time series from nine precipitation stations situated in the study area or its close proximity were used in the analysis (Fig. 2). The short time series included five years of measurements from 2001 to 2005.

Groundwater level measurements from 56 piezometers (Fig. 2) for an observation period 1998–2007 have been used in the analysis. The groundwater measurement network of the KNP, designed in 1995, included piezometers located along the longitudinal transects crossing both wetland belts and the sand dune zones; the measurements were taken with a 14-day interval (Krogulec, 2003b). All these data were recalculated into monthly values to be compatible and also better suit the possible precision of the calculations. The joint observation period for precipitation and groundwater is five years from 2001 to 2005.

Both precipitation and groundwater time series were subject to data quality controls and field inspection of the observation stations was carried out, which resulted in elimination of one doubtful groundwater series and one precipitation series.

An exploratory analysis of precipitation and groundwater time series with respect to their patterns of variability in space and time was performed in search for common patterns across space between variables. Special tasks were identification of possible non-stationarity in fluctuations, as well as deviating fluctuation patterns that may be traced to human impact. Visual analyses and statistical analysis applying the Principal Component Analysis (Jolliffe, 1990) allowed identifying roughly three spatial groups with different average depth of the groundwater. Most of observation points in the group with deepest groundwater are located in the dune areas (average level 2.5 m), while two groups with shallow (average level 0.8 m) and intermediate (average level 1.1 m) groundwater levels are located in wetlands.

The same statistical method was applied to explore spatio-temporal pattern of precipitation. The temporal fluctuation patterns at all stations was almost identical and the difference between the average precipitation volumes among the stations was minor. The series length was too short to allow a strict trend analyses but a visual inspection of the plots of the series indicated no trends in precipitation for the studied period.

A simple water balance model WASMOD (Xu, 2002) was applied and calibrated with monthly data on precipitation, temperature and runoff as input data (from Somorowska et al., 2011) for the Canal Łasica catchment. The model simulation yielded estimated time series of rain and snowmelt, and actual evapotranspiration. The strong seasonal variation in actual evapotranspiration appeared to be well reflected in the seasonal variation of observed groundwater depths.

3.1.3. Soil and vegetation maps

Two soil maps of the study area, one showing primary soil cover (the one that used to exist a few decades ago) and one showing the present soil cover, created by Piórkowski et al. (2011) were used. Table 1 presents the soil types present in the maps and their codes that will be used hereinafter. Soil types are ordered according to

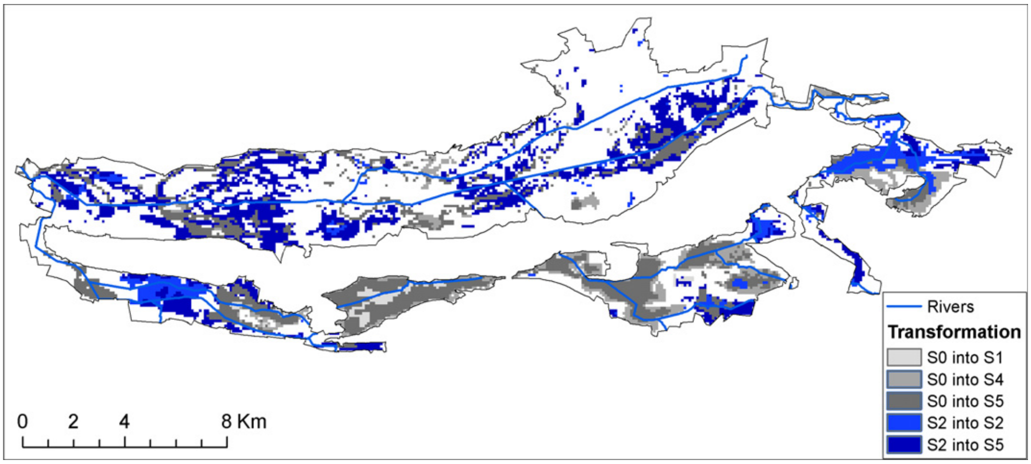

Fig. 3. Map of different types of soil transformation from primary to secondary soils.

the decreasing wetness state according to description of soils in Piórkowski et al. (2011).

The overlay of two mentioned maps facilitated analysing transformation (dynamics) of soil cover over the last few decades. Of particular interest were those areas which used to have wet soil cover in the past (S0 and S2) and have recently turned into drier classes from Table 1. In particular, three types of transformations of primary peat soils were observed:

1. S0 into S1; this class includes moderately degraded peat soils;
2. S0 into S4; this class includes degraded peat soils;
3. S0 into S5; this class includes heavily degraded peat soils;

Two types of transformations of primary muddy-gley soils were observed:

1. S2 into S2; this class includes muddy-gley soils in balance;
2. S2 into S5; this class includes degraded muddy-gley soils.

Fig. 3 illustrates all of the above combinations. As discussed in Piórkowski et al. (2011), soil deterioration has been more profound in the southern wetland belt, which is visible on the map.

A detailed vegetation map of the study area was described in Michalska-Hejduk et al. (2011). For the purpose of this study, a generalised map, reported in Okruszko et al. (2011) was used. Classification of vegetation was similar to the one of soils. Three types were established: wet communities, moist communities and mesic and dry communities. A classification of the state of vegetation communities in terms of being in balance or degraded and estimation of a possibility of restoration of wet communities was also available (Olech et al., 2011). Combining this information with the available vegetation map, a map of the state of vegetation communities was created (Fig. 4). Table 2 presents the names of the distinguished vegetation communities and their state, estimating also a possibility of restoration. Fig. 4 shows wet communities only, as they are of primary interest in this study.

Table 1
Soil types and codes with description of their wetness state and percent area occupied by each soil in the currently observed soil cover of the study area (after Piórkowski et al., 2011).

Soil code	Soil type	Wetness state	Symbol	Percent area
S0	Peat	Wet	++	0%[a]
S1	Peat-mud	Moderately wet	+	1.1%
S2	Muddy-gley	Moderately wet	+	4.7%
S3	Gley	Neutral	0	4.9%
S4	Muddy-muck	Moderately dry	–	15.6%
S5	Mineral-moorsh	Dry	– –	25.4%
S6	Mucky soil	Very dry	– – –	41.1%
S7	Mineral soil	n/a[b]	n/a	7.1%

[a] Peat soils (S0) have degraded over the recent decades and they are no longer present in the soil cover of the study area. They are kept in the table because they are considered as "primary" soils, whose transformations into other soil types are further studied.
[b] Mineral soils (S7) are non-hydrogenic soils, hence not of interest within this paper.

3.2. Methods

Two basic tasks have been distinguished:

1. analysis of cause–effects relations;
2. assessment of environmental changes.

The effect variables, i.e. vegetation cover and soil types, have both relatively long temporal scale of variability, the latter significantly the longest, in the order of decades, but there are few data to objectively describe variation of vegetation and soils over time. The length of the climatic and hydrologic records available was relatively short in comparison, which makes it impossible to identify cause–effect relations from time series analysis only. Hence, this identification had to rely on spatial data only, i.e. maps described in Section 3.1.

The method for analysing these data needs to bridge over research results from purely quantitative relations and classifications to mathematical modelling and statistical analysis. This creates specific problems, for example, in relation to:

Table 2
Classes of vegetation communities distinguished (adapted from Olech et al., 2011).

Code	Community type	State	Restoration of wet conditions	Percent area
V1	Wet	In balance	Not necessary	20.4%
V2	Wet	Degraded I	Advisable	9.6%
V3	Wet	Degraded II	Difficult	1.0%
V4	Moist	In balance	Not necessary	7.1%
V5	Moist	Degraded I	Advisable	4.5%
V6	Moist	Degraded II	Difficult	9.6%
V7[a]	Mesic and dry	In balance	Not recommended	26.0%
V8[a]	Mesic and dry	Degraded I	Advisable	18.0%
V9[a]	Mesic and dry	Degraded II	Difficult	3.8%

[a] V7 are natural or semi-natural communities, while V8 and V9 are anthropogenic communities.

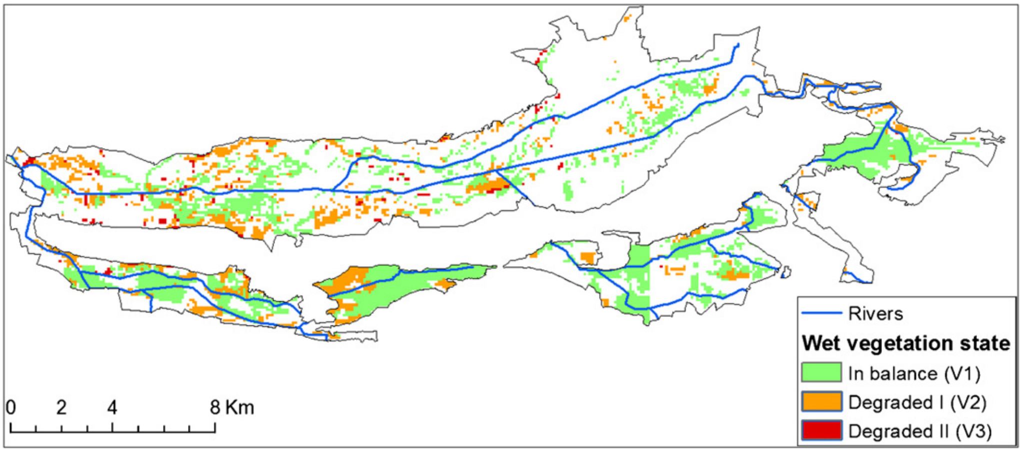

Fig. 4. Map of classes of wet vegetation communities established based on the evaluation of their state.

– different methods used for interpolation from observations sites to grids;
– combining indicator maps (classes) with maps of "continua" (groundwater levels);
– consistent estimation of uncertainty.

To overcome the problem with vital differences in the character of input data the basic spatial analysis is performed in GIS frame with grid cells of the size 100 m × 100 m as the basic units a_k, $k = 1, \ldots, M$, where M is the total number of grid cells in the study. This required a transformation of original soil and vegetation vector maps into 100 m resolution raster and interpolation of groundwater level data to the regular grid net applying stochastic interpolation – kriging.

Variables defined for grid cells are categorised into cause variables X_k and effect variables Y_k. They both take on discrete values, say, $X_k = 1, 2, \ldots, K_X$ or $X_k = -1, 0, 1$. K_X in the first case is the number of classes for variable X. The second case is an example of classification according to wetness classes dry (−1), intermediate (0) and wet (1). Stochastic interpolation methods directly provide estimation errors so that the uncertainty in such classified estimates can be determined in terms of probabilities for correct or incorrect classification. Other types of classified data contain manual evaluation of field survey data, for which it is more difficult to assign uncertainty measures.

The basic tool for identifying cause–effect relations from these maps were contingency tables. Comparing the patterns of variability across space of the effect variables, i.e. vegetation cover and soil types, and these of the causal variables, like amount of precipitation, groundwater depth and human impacts it is possible to identify cause–effect relations, assuming that the effect variables are in balance with the causal variables, as the data available do not allow making this identification from spatio-temporal data with records extending over the characteristic scales of variability. Further analysis anyhow allows verifying this assumption.

A contingency table may be used to record and analyse the relation between the two variables X and Y. It displays the number of counts n_{ij} across all grid cells a_k, $k = 1, \ldots, M$ for which $X_k = i$ and $Y_k = j$ in a matrix format $\mathbf{N}_{XY}$:

$$\mathbf{N}_{XY} = \begin{pmatrix} n_{11} & n_{12} & \cdots & n_{1N_Y} \\ n_{21} & n_{22} & \cdots & n_{2N_Y} \\ \vdots & \vdots & \vdots & \vdots \\ n_{N_X1} & n_{N_X2} & \cdots & n_{N_XN_Y} \end{pmatrix} \quad (1)$$

If the numbers n_{ij} in the different columns vary significantly between the rows (or vice versa), there is a contingency between the two variables. In other words, the two variables are not independent. If there is no contingency between the two variables, the two variables are independent. It can be convenient to normalise the matrix values in relation to the total number of observations, i.e. $n'_{ij} = n_{ij}/M$. These normalised values can be interpreted as empirical probabilities that the joint class $X = i$ and $Y = j$ is observed.

As mentioned earlier, the identification of cause–effect relations has to rely on spatial data only due to the lack of time series for the effect variables. Cause–effect relations will be identified from a successive search of contingency between cause and effect variables. In case contingency is confirmed, the contingency table can be transformed to a matrix of conditional probabilities of observing a certain effect given a certain cause, which opens for probability predictions of possible environmental change induced by change in predictors (causes).

The contingency table normalised with respect to the total number of observations gives estimates of absolute probabilities of observing a certain combination (i,j) of cause and effect variables. Dividing all row elements with respect to the total number of the observed elements in the row M_i, yields empirical probabilities $p_{ij} = n_{ij}/M_i$ of observing the effect Y_j given the cause X_i, i.e. $p_{ij} = P(Y = j \mid X = i)$, which is expressed in terms of the conditional probability matrix $\mathbf{P}_{XY}$:

$$\mathbf{P}_{XY} = \begin{pmatrix} p_{11} & p_{12} & \cdots & p_{1N_Y} \\ p_{21} & p_{22} & \cdots & p_{2N_Y} \\ \vdots & \vdots & \vdots & \vdots \\ p_{N_X1} & p_{N_X2} & \cdots & p_{N_XN_Y} \end{pmatrix} \quad (2)$$

The probabilities in the rows sum up to one $\sum_j^{N_Y} p_{ij} = 1; \quad i = 1, \ldots, N_X$.

The matrix $\mathbf{P}_{XY}$ will be the tool for making probability predictions of environmental change in Y for a given characterisation of state probabilities in X after an alteration. The starting point is thus the absolute probabilities $p_i^{(X)}; \quad i = 1, \ldots, N_X$ of observing X in a certain state (category) i. The prediction equation forthe absolute

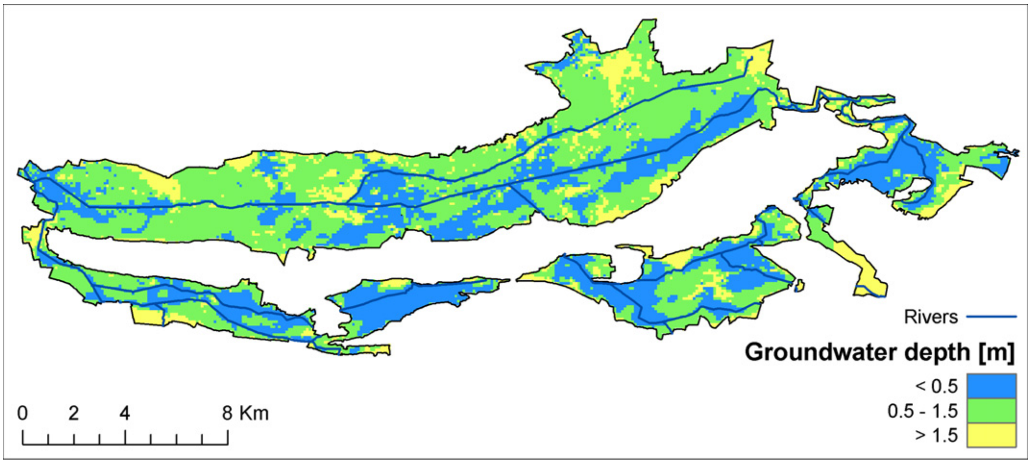

Fig. 5. Map of mean groundwater depth for the study area.

probabilities $p_j^{(Y)}$; $i = 1, \ldots, N_Y$ of observing Y in a certain state (category) j is:

$$\mathbf{P}^{(Y)} = \mathbf{P}^{(X)}\mathbf{P}_{XY} = (p_1^{(Y)}, p_2^{(Y)}, \ldots, p_{N_Y}^{(Y)}) = (p_1^{(X)}, p_2^{(X)}, \ldots, p_{N_X}^{(X)}) \begin{pmatrix} p_{11} & p_{12} & \cdots & p_{1N_Y} \\ p_{21} & p_{22} & \cdots & p_{2N_Y} \\ \vdots & \vdots & \vdots & \vdots \\ p_{N_X1} & p_{N_X2} & \cdots & p_{N_XN_Y} \end{pmatrix} \quad (3)$$

where $\boldsymbol{P}^{(X)}$ and $\boldsymbol{P}^{(Y)}$ are row vectors of state probabilities of X and Y, respectively.

4. Results and discussion

4.1. Analyses of present cause–effect relations

The analysis started with studying variability of precipitation and groundwater data, for which time series of limited length were available, looking for common patterns and identification of eventual non-stationarity already referred to in Section 3. The subsequent analyses of cause–effect relations relies on spatial data only, i.e. digital map data and groundwater depth maps. Firstly, the correlations between soil type and vegetation class on the one hand and groundwater depths on the other hand are studied, and secondly, the dependencies between dynamic characteristics of soil and vegetation and groundwater depths.

As already mentioned, groundwater level observations (mean values from the time period 2000–2008) were interpolated in space using stochastic interpolation – kriging. For further investigation it was not the groundwater levels that were of primary interest but the groundwater depths. These were estimated by subtracting the estimated groundwater levels from the digital elevation maps. Groundwater depths were categorised in three intervals <0.5 m (saturated moisture conditions prevail due to shallow groundwater), 0.5–1.5 m (intermediate situation) and >1.5 m (groundwater is deep and hardly controls soil moisture conditions). The resulting map is shown in Fig. 5. The first and second classes are dominating in the study area. Table 3 presents the statistics of the interpolated groundwater depths.

The standard errors of the interpolated values vary significantly across space, being largest in areas with no observations. With the assumption of normally distributed errors and a known characteristic standard error ($\sigma_e = 0.33$ m) the probability density functions $f_Z(z)$ and cumulative distribution functions $F_Z(z)$ of estimated depths z can be estimated. In the case of three groundwater wetness classes, namely <0.5 m, 0.5–1.5 m, and >1.5 m the resulting distributions are shown in Fig. 6. These distributions can be used to evaluate a transition matrix of probabilities p_{ij} that the true class i will be classified as j (Table 4). The diagonal (bold values) in this matrix gives the probabilities of a correct classification.

The most apparent result of the uncertainty in the stochastic interpolation procedure is the fact that the interpolated values of the groundwater level can be higher than the land surface, as determined from topography (it should be noted that the topographical map as such also contains errors, which we for the time being neglect). This results in negative groundwater depths (note that 5% quantile in Table 3 yields −0.26 m). The area with negative mean groundwater depth estimated from kriging is about 11% of the total area. The negative depths are thus fully explained by the uncertainty related to the present density of groundwater observations.

Table 3
Statistics of stochastically interpolated (kriged) groundwater depths.

Parameter	Estimates
Mean [m]	0.83
Median [m]	0.73
Mode [m]	0.66
Standard deviation [m]	0.85
Skewness	2.05
5% Quantile [m]	−0.26
95% Quantile [m]	2.22

Table 4
Classification probabilities of groundwater depth for different estimations of the characteristic standard error.

p_{ij}	Groundwater class			
	<0	0–0.5	0.5–1.5	>1.5
σ_e = 0.34 m (mean)				
0–0.5	0.25	**0.50**	0.24	0.00
0.5–1.5	0.01	0.12	**0.73**	0.13
>1.5	0.00	0.00	0.02	**0.98**
σ_e = 0.32 m (median)				
0–0.5	0.24	**0.52**	0.24	0.00
0.5–1.5	0.01	0.12	**0.75**	0.13
>1.5	0.00	0.00	0.02	**0.98**
σ_e = 0.24 m (mode)				
0–0.5	0.19	**0.62**	0.19	0.00
0.5–1.5	0.00	0.09	**0.81**	0.1
>1.5	0.00	0.00	0.01	**0.99**

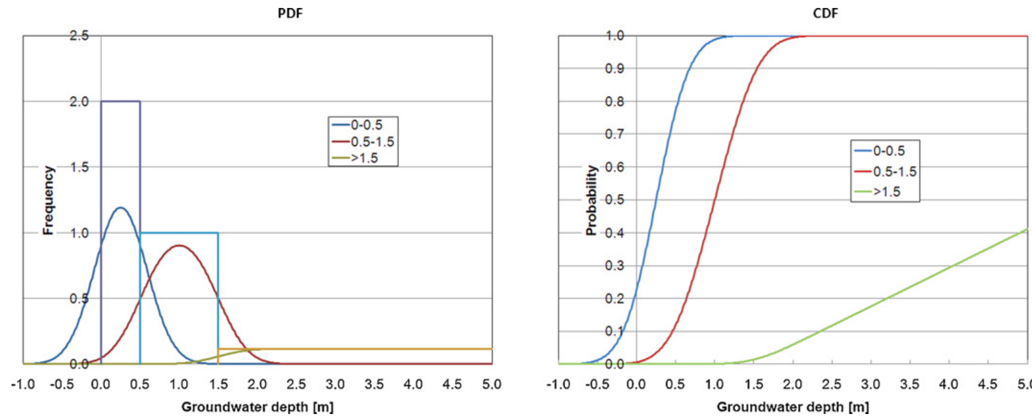

Fig. 6. Probability density functions $f_Z(z)$ (left graph) and cumulative distribution functions $F_Z(z)$ (right graph) of estimated depths z for three groundwater wetness classes. The background rectangles in the left graph show the uniform distributions for the respective depth class for the case with no errors. The characteristic standard error is $\sigma_e = 0.33$ m.

Mean groundwater depths appeared to be overall well in accordance with the soil wetness classes (cf. Table 1), as evident from Fig. 7A, with an exception of gley soils (S3), which should have been the 5th in this hierarchy and not the 3rd. This could be due to uncertainties in map information. The difference between the wettest and driest soil is of higher magnitude and equals approximately 1.5 m.

Mean groundwater depths for different vegetation types (Fig. 7B) are also well in accordance with the vegetation wetness hierarchy (Table 2). The difference between the wet and mesic and dry vegetation is of lower magnitude than in the case of two most distant types of soils and is approximately 0.55 m.

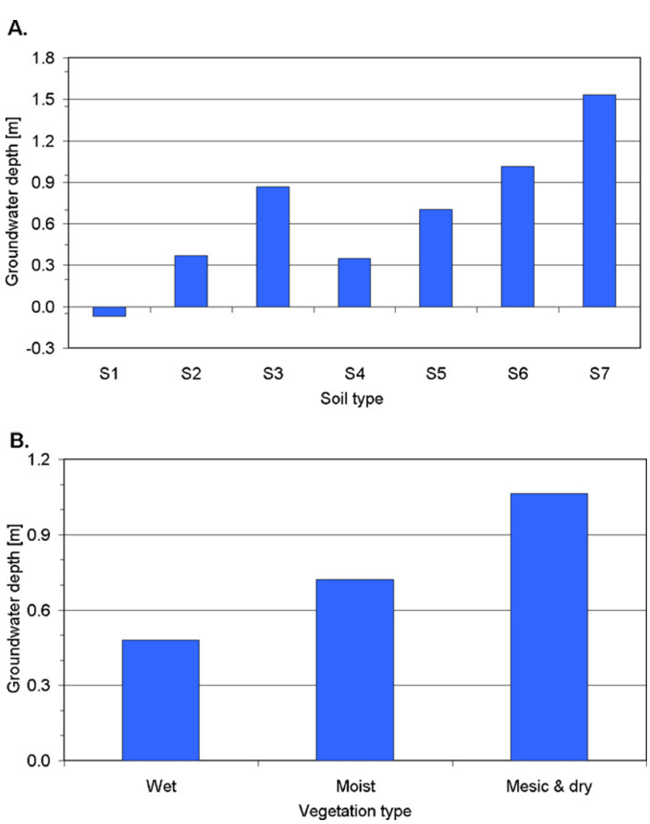

Fig. 7. Mean groundwater depth in different soil (A) and vegetation (B) types.

There is a spatial correlation between the occurrence of three types of wet vegetation in different state (V1, V2 and V3; cf. Table 2) and soil wetness classes S1–S7. For example, we observed that wet vegetation in balance (V1) grows predominantly on soil type S3 (40%), followed by type S6 (35%) and S1 (11%). In contrast, the frequency of soil types within two classes of degraded wet vegetation (V2 and V3) is different: within the class V2, soil type S6 dominates (49%), followed by the types S3 (32%) and S1 (9%); within the class V3 the highest frequency can be observed for soil type S5 (45%), followed by the types S6 (36%) and S4 (11%). Hence, the more degraded wet vegetation is, the higher is the frequency of soil types characterised by limited moisture conditions.

It can be concluded that the wetness state of soils (with the exception of gley soils) and vegetation, respectively, is well in accordance with the groundwater depth, which allows using this latter as a predictor.

4.2. *Predicting change in a wetland environment*

The cause–effect relations can be presented in terms of contingency tables that can be further elaborated for prediction purposes. The cause variable will serve as predictor (independent variable) and the effect as predictand (dependent variable). All data represent classes, which can be characterised in terms of integer values and we interpret them as outcomes of discrete random variables. It has been noted that the contingencies are all the events that could possibly happen. In the present application this means what we are able to observe with the available spatial data. We will not be able to predict something that is not observed at present.

The point of departure is thus contingency tables based on the gridded map information in accordance with Eq. (1). Such table shows the counts of how many times each of the contingencies actually happened in a particular sample, which has been graphically illustrated in the previous chapter. The groundwater state probabilities will be used as predictors with the aim of predicting the state probabilities of vegetation in accordance with the classes introduced earlier (cf. Fig. 4 and Table 2). Two different resolutions of groundwater depths will be used. In the first one four classes are defined: <0 m, 0–0.5 m, 0.5–1.5 m and >1.5 m. The negative groundwater depths resulting from the uncertainty in the interpolation procedure should be also considered. They are treated as a separate class, which should be interpreted as areas with very shallow groundwater depth or open water. Table 5 shows the contingency table for this case. The second classification is still finer, with six

Table 5
Contingency table of groundwater depth class (<0 m, 0–0.5 m, 0.5–1.5 m, >1.5) and state of vegetation.

Code	Community type	State	Groundwater class (state)				Sum
			<0	0–0.5	0.5–1.5	>1.5	
V1	Wet	In balance	952	1221	1326	118	3617
V2	Wet	Degraded I	200	482	799	133	1614
V3	Wet	Degraded II	24	47	89	9	169
V4	Moist	In balance	146	321	693	58	1218
V5	Moist	Degraded I	150	200	380	32	762
V6	Moist	Degraded II	121	463	843	225	1652
V7	Mesic and dry	In balance	193	677	2756	885	4511
V8	Mesic and dry	Degraded I	241	535	1767	649	3192
V9	Mesic and dry	Degraded II	19	84	324	212	639
Sum			2046	4030	8977	2321	17,374

Table 6
Contingency table of groundwater depth class (<0 m, 0–0.5 m, 0.5–1 m, 1–2 m, 2–4 m, >4 m) and state of vegetation.

Code	Community type	State	Groundwater class (state)						Sum
			<0	0–0.5	0.5–1	1–2	2–4	>4	
V1	Wet	In balance	952	1221	928	472	40	4	3617
V2	Wet	Degraded I	200	482	527	338	64	3	1614
V3	Wet	Degraded II	24	47	44	53	1	0	169
V4	Moist	In balance	146	321	465	261	25	0	1218
V5	Moist	Degraded I	150	200	240	158	14	0	762
V6	Moist	Degraded II	121	463	529	411	106	22	1652
V7	Mesic and dry	In balance	193	677	1663	1590	317	71	4511
V8	Mesic and dry	Degraded I	241	535	1049	1064	235	68	3192
V9	Mesic and dry	Degraded II	19	84	166	268	90	12	639
Sum			2046	4030	5611	4615	892	180	17,374

classes: <0 m, 0–0.5 m, 0.5–1 m, 1–2 m, 2–4 m, >4 m. Contingency for this case is presented in Table 6.

Analysing the individual counts of the two tables leads to the conclusion that it is only in the wet vegetation communities (hereinafter referred to as *wetlands* for simplicity) that a significant correlation between the groundwater depth and the vegetation can be seen. For other vegetation communities other factors than wetness conditions play a major role. Hence, only in wetlands it is possible to use the groundwater conditions as a predictor.

From the contingency tables the vectors $\mathbf{P}^{(X)}$ of absolute state probabilities of groundwater for the above two groundwater depth classifications are estimated by dividing the number of counts of groundwater depth in a certain class by the total number of counts for the wetlands. The calculation yields the estimates:

$$\mathbf{P}^{(X)} = (0.218, 0.324, 0.410, 0.048);$$
$$\mathbf{P}^{(X)} = (0.218, 0.324, 0.278, 0.160, 0.019, 0.001);$$

The elements of the matrix of estimated conditional probabilities $\mathbf{P}_{XY}$ (Eq. (1)) are shown in Tables 7 and 8.

The vector of absolute state probabilities for wetland vegetation is estimated from the contingency table to:

$$\mathbf{P}^{(Y)} = (0.670, 0.299, 0.031);$$

These vectors and matrices satisfy the prediction Eq. (3). We also have the necessary information to predict changes in the vegetation state probabilities in response to changing state probabilities of groundwater. The groundwater depth is not a constant within a certain community area but shows variations around a mean depth that follow a lognormal distribution. The statistical analysis of the piezometer data in Table 3 indicated a standard deviation of around 0.85 in wetlands. Fig. 8 shows how the state probabilities for the different depth classes of groundwater change with the average depth of groundwater for this coefficient of variation. These theoretical probabilities include the classification uncertainties of groundwater. With application of the prediction equation (Eq. (3)) and the known conditional probability matrices (Tables 7 and 8) it is now possible to estimate the change in the state probabilities of the vegetation. The result is shown in Fig. 9.

The graphs show a very strong influence of the mean groundwater level on the state of vegetation between that “in balance” and “degraded”.

As already noted, it is not possible to predict something that is not observed at present with the suggested approach. The area occupied by the class V3 “degraded II” is very small, and therefore the detected effect of groundwater depth change on degradation class is insignificant. Degraded areas are possibly caused by other

Table 7
Probabilities (p_{ij}) for degree of degradation conditioned on groundwater depth class (0–0.5 m, 0.5–1.5 m, >1.5).

Groundwater state	Wetland state		
	In balance (V1)	Degraded I (V2)	Degraded II (V3)
<0	0.81	0.17	0.02
0–0.5	0.74	0.23	0.02
0.5–1.5	0.60	0.36	0.04
>1.5	0.45	0.51	0.04

Table 8
Probabilities (p_{ij}) for degree of degradation conditioned on ground water depth class (0–0.5 m, 0.5–1 m, 1–2 m, 2–4 m, >4 m).

Groundwater state	Wetland state		
	In balance (V1)	Degraded I (V2)	Degraded II (V3)
<0	0.81	0.17	0.02
0–0.5	0.74	0.23	0.02
0.5–1.0	0.62	0.36	0.03
1.0–2.0	0.55	0.40	0.06
2.0–4.0	0.38	0.61	0.01
>4.0	0.57	0.43	0.00

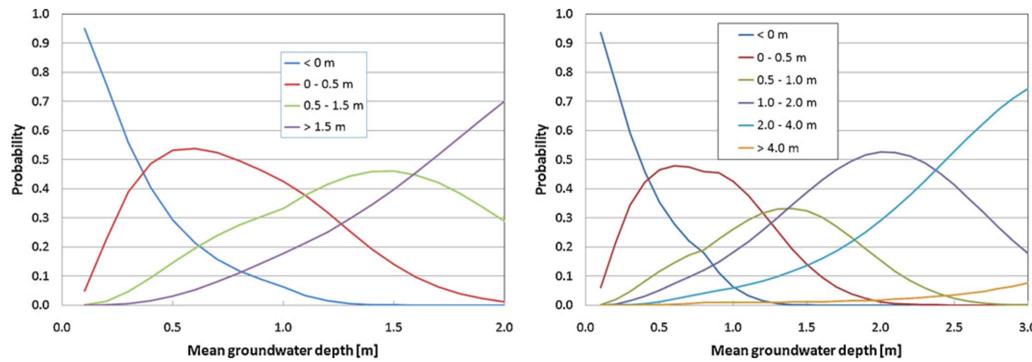

Fig. 8. Dependence of state probabilities for groundwater classes on mean groundwater depth for wetlands. The lognormal distribution is assumed with a standard deviation equal to 0.85.

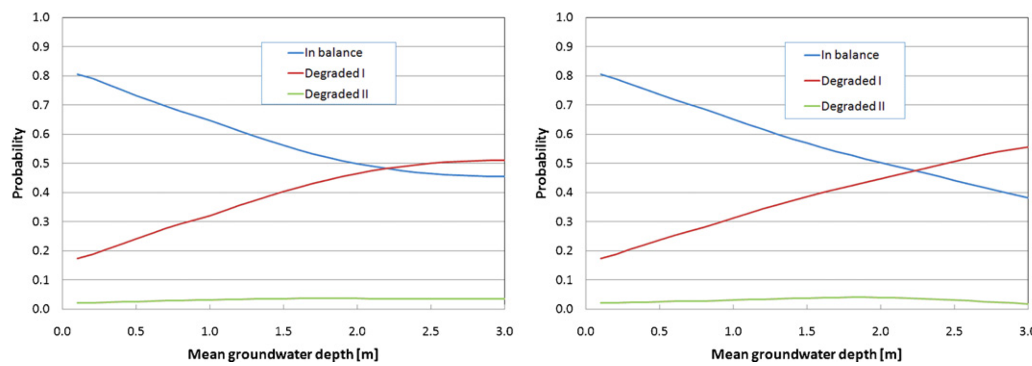

Fig. 9. Change in the state probabilities of the vegetation as a function of mean groundwater depth for wetlands for the case with four and six groundwater classes.

factors and not linked to wetness conditions only. The present available information does not allow identification of these causes.

The analyses presented earlier indicated that other reasons than decreasing groundwater table might be the cause of degradation of the vegetation in particular in moist, mesic and dry communities in KPN. It is well-known that peatland degradation is most often caused by human activity (Van Diggelen et al., 1991), and in particular, construction of drainage system to gain land for agricultural production, mainly for meadows and pastures. Shorter storage of water on the surface and decreased moisture of organic soil lead to peat mineralisation and degradation. Vegetation changes from highly valuable and biodiverse ecosystems to monoculture meadows and pastures have been shown in many studies (e.g. Van Diggelen et al., 1991; Olde Venterink et al., 2009; Klimkowska et al., 2010a,b; Tahvanainen, 2011).

No quantitative information about the drainage activities in KPN during the two last centuries or water use (e.g. groundwater levels before these works; statistics over changes in land use for the time period since the drainage works started; human and agricultural water consumption, etc.) was available. To compensate for the absence of any information on anthropogenic activities a map of the density of drainage ditches and a map of the anthropogenic land uses based on CORINE Land Cover 2006 were created. However, no clear relationships have been found based on these maps. Given the major role of human activities in degradation of the soil and vegetation in general, an inventory of all historical and present human activities within KPN can be strongly recommended.

The results of the analyses presented here indicate a very strong influence of the mean groundwater level on the state of wet zone vegetation between what is “in balance” and “degraded”. This opens an opportunity for evaluating the impact of changes in groundwater depth (e.g. as a result of closure of some drainage channels) on the state of vegetation. Five sets of scenarios for changing groundwater conditions presented in Table 9 were assessed. The scenarios have been obtained by means of the MODFLOW model (for details see Okruszko et al., 2011).[1]

We will here utilise the difference in statistics in the different scenarios in relation to the present situation to predict the potential impact on the vegetation state. The statistics of present

Table 9
Five wetland restoration scenarios modelled with MODFLOW (from Okruszko et al., 2011).

Scenario 1	Exclusion of the whole existing artificial drainage system including the Canal Łasica by filling up all the ditches and canals. This scenario corresponds to restoration of pre-drainage conditions
Scenario 2	Urbanisation of the KNP surroundings expressed by intensification of groundwater uptake from the current level of 4.6 m^3/day/km^2 by 100%
Scenario 3	Restoration of meanders in the Canal Łasica expressed by a reduction of the channel filtration resistance, which should correspond to the first phase after restoration[a]
Scenario 4	Restoration of meanders in the Canal Łasica expressed by an increase of the channel filtration resistance, which should correspond to the second phase after restoration[a]
Scenario 5	Damming the River Vistula causing an average increase of its water level by 2 m

[a] The results coming from Scenarios 3 and 4 should be analysed qualitatively rather than quantitatively due to simplifying character of the model assumptions.

[1] Statistics of the present groundwater depths modelled by MODFLOW and those interpolated by kriging were within the error accuracy limits for kriging and mathematical modelling. For details see Gottschalk et al. (2011).

Table 10
Statistics of groundwater depths for the five different scenarios of groundwater in the wetland zone and that of the present situation as determined by groundwater modelling.

Parameter	Present	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Mean [m]	0.69	0.36	0.70	0.78	0.59	0.66
Median [m]	0.61	0.31	0.62	0.72	0.49	0.58
Mode [m]	0.53	0.24	0.51	0.66	0.39	0.40
Standard deviation [m]	0.77	0.85	0.77	0.76	0.79	0.75
Skewness	1.87	1.40	1.86	1.84	1.81	1.98
5% Quantile [m]	−0.30	−0.84	−0.30	−0.23	−0.44	−0.31
95% Quantile [m]	1.99	1.83	2.01	2.05	1.96	1.90
% Negative values	13.1	32.2	13.0	10.1	19.1	13.7

Table 11
Expected changes in vegetation states in hectares by changing groundwater conditions for the five different scenarios.

Vegetation states		In balance	Degraded I	Degraded II	Sum
Present state [ha]		3618	1620	162	5400
Expected	Scenario 1	+333	−310	−24	0
change	Scenario 2	−12	+11	+1	0
in	Scenario 3	−99	+92	+7	0
state	Scenario 4	+112	−104	−8	0
[ha]	Scenario 5	+36	−34	−2	0

groundwater depths obtained by means of MODFLOW and those for the five scenarios are given in Table 10.

For the prediction we only utilise the relative changes in the statistics between the modelled present situation and that of the scenarios. With application of the prediction equation (Eq. (3)) and the known conditional probability matrices (Tables 7 and 8) it is possible to estimate the change in the state probabilities of the vegetation for the different scenarios in the same way as in the previous section. The result is shown in Table 11 in terms of the expected value of the area that will change its vegetation state in response to the changed groundwater conditions.

The improvement, i.e. the change from a degraded state to the one in balance in response to different scenarios is illustrated in Fig. 10. It is obvious that only groundwater depth changes in the Scenario 1 can be expected to have some significant positive effects on the vegetation state. This statement naturally contains uncertainty and that is why such statements by necessity should involve probability measures and expected values.

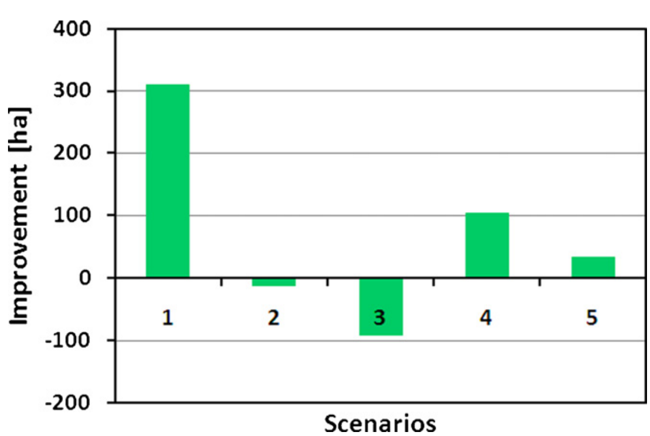

Fig. 10. Predicted improvements (the change from a degraded state to the one in balance) for five different scenarios (cf. Okruszko et al., 2011). Scenario 1 assumes exclusion of the whole existing drainage system; Scenario 2 assumes intensification of groundwater uptake; Scenarios 3 and 4 assume reduction/increase of the channel filtration resistance; Scenario 5 assumes the increase of the water level in the River Vistula.

5. Conclusions

Assessing degradation of wetlands is a difficult task that has many facets. Wetlands are complex systems, where all the components (hydrological conditions; flora and fauna) are interrelated. Meanwhile, very often each of the system components is studied separately, which yields valuable information about it but leaves out the interrelationship. Degradation of wetlands, on the other hand, signifies disturbances in balance among system components. Thus, understanding cause–effect relations is a necessary premise for testing and implementation of restoration measures. Herein an approach that synthesises the findings on separate elements of the wetland system of KPN has been elaborated and used to predict an impact of restoration measures on the state of vegetation there.

The exploratory data analyses gave no evidence of trends for the relatively short groundwater and climatic records at disposal nor direct human impact at present. Fluctuation of monthly groundwater depths appeared to be stronger influenced by the monthly evapotranspiration from the vegetation cover than by relative fluctuations in rain and snowmelt amounts. Only in wetland areas the degree of degradation of vegetation and soils could be related to groundwater conditions. In other areas other causal factors may be important, which indicates a necessity of analysing also other types of information (e.g. socio-economic) that was not available for our study.

Possible cause–effect relationship with respect to one of a manifold of factors that can be behind the observed deterioration, namely—changing groundwater levels, have thus been examined. Groundwater observations that were available only for a relatively short period were interpolated to grid cells, which made this information compatible to available map information on the state of the soils and vegetation with respect to their respective degradation.

A special attention was paid to uncertainty involved when using map and other qualitative information originating from different sources. The prediction errors in the maps of interpolated groundwater levels are relatively small and depend on the density of observations, which as such is very high. Possible estimation errors have been considered in the prediction model. No measure of uncertainty in soil and vegetation mapping was available to consider them in the prediction model.

The strong dependence between soil and vegetation state and groundwater average depth revealed by the analyses was used for probability prediction purposes based on four different scenarios of changes in groundwater depths. Given all the uncertainties involved, a prediction involved probability measures and expected values. A probability prediction of the impact on the vegetation state for different scenarios used indicated that only amelioration according to scenario 1 could significantly improve present conditions. A limitation of the approach in the present application is that it can only predict events contained in the available spatial data and it is not possible to predict something that is not observed at present. A broader application of the approach calls for multiple studies in different wetland conditions to broaden the data sample to include wider range of vegetation and soil conditions.

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