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Environmental water quantity projections under market-driven and sustainability-driven future scenarios in the Narew basin, Poland

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Abstract
The aim of this paper is to assess the impact of four scenarios combining changes in climate, atmospheric CO_2, land and water use possible by 2050 on the specific set of ecologically-relevant flow regime indicators that define environmental flow requirements in a semi-natural river basin in Poland. This aim is presented through a modelling case study using Soil & Water Assessment Tool (SWAT). Indicators show both positive and negative responses to future changes. Warm projections from IPSL-CM4 GCM combined with sustainable land and water use projections (SuE) produce the most negative changes, while warm and wet projections from MIROC3.2 combined with market-driven projections (EcF) the most positive changes. Climate change overshadows land and water use change in terms of the magnitude of projected flow alterations. The future of environmental water quantity is brighter under market-driven rather than sustainability-driven scenario, which shows that sustainability for terrestrial ecosystems (e.g. more forests and grasslands) can be at variance with sustainability for riverine and riparian ecosystems (requiring sufficient amount and proper timing of river flows).

Key words environmental flows, SWAT, river ecosystem, climate change, land use change, future scenarios

1 INTRODUCTION

The discharge of water exercises an important control over riverine ecosystems, along with temperature and nutrient availability (Moss 2010). River ecosystems are adapted to the flow regime. The natural flow paradigm (Poff *et al.* 1997) takes the natural system as its starting point and argues that the flow regime of a river, comprising the five key components of magnitude frequency, duration, timing, rate of change and overall variability, is central to sustaining biodiversity and ecosystem integrity. However, flow regimes are not stationary, but alter as climate changes (Palmer *et al*. 2009) or water is managed for public supply, irrigation or hydropower production (Nilsson *et al.* 2005). Indicators of the hydrological regime and ecosystem response are needed to understand how river systems may change and to help us take any actions necessary, if we want to avoid environmental degradation. This is particularly important in Europe where implementation of the Water Framework Directive requires member states to assess the Ecological Status in all water bodies, which is measured in terms of deviation from natural reference conditions (Acreman and Ferguson 2010).

A wide range of water management indicators is available, such as those relating to irrigation water needs, flood risk or pollution level and the appropriateness of the indicator will depend on the geographical setting and river basin planning objectives. In our view, in near-pristine river basins it is important to focus on the indicators related to basic ecosystem requirements since biodiversity conservation is the main objective (Acreman *et al*., in review). One approach to defining indicators is to identify the fundamental building blocks of the flow regime that influence key elements of the river ecosystem (King *et al*. 2000). This method was proposed for defining appropriate flow releases from dams in the UK to achieve Good Ecological Potential in downstream river water bodies to fulfil the European Water Framework Directive (Acreman *et al*. 2009). It was further developed to quantify the most important (for biota), catchment-specific blocks/components of the flow regime in the Narew River (Piniewski *et al*. 2011) and has been applied to defining releases from hydropower dams in Norway (Alfredson *et al*. 2012). The approach lends itself well to design of indicators of hydro-ecological alteration.

Stressors are commonly divided into natural (i.e. climatic) and human-induced (anthropogenic) – although we now believe climate is being altered indirectly by man's activities. Global warming, driven by increased greenhouse gas emissions, has been observed for decades and reported in various global (IPCC 2001, 2007) and Polish (Kundzewicz and Matczak 2012) studies. In Poland, in particular an air temperature rise have been reported since 1950s (Degirmendzic *et al*. 2004), and an increase in the ratio of precipitation in the cold period to precipitation in the warm period, as well as an increase in the number of dry days in a year (Pińskwar 2010). Observed patterns of climate change in Poland have been indicated by some researchers (Kundzewicz and Matczak 2012) as the cause of more frequent extreme hydrological events - both floods and droughts – today than in the past.

Land use is also changing at all spatial scales and is a key component of global change affecting ecological systems (Vitousek 1994). Lambin *et al*. (2001) have emphasised four major classes of global land use change: tropical deforestation, rangeland modifications, agricultural intensification and urbanization. The latter has been significant in Poland, where for example, the built-up areas in the Warsaw metropolitan district have increased by 63% over the period 1950-1990, mainly through conversion of arable land (Solon 2009). More recent patterns of land use change in Central and Eastern Europe have been largely related to the transition of state-command to a market-driven economy in the last decade of the twentieth century. Prishchepov *et al*. (2012) demonstrated that the rate of agricultural land abandonment during the first decade of transition reached 14% in the NE Poland, but was significantly lower than in the former Soviet Union countries (e.g. 42% for Latvia). Catchment land use change and associated water resource development inevitably lead to changes in one or more aspects of the flow regime resulting in alterations in species and biological communities and often declines in aquatic biodiversity (Poff et al., 1997; Bunn & Arthington, 2002, Arthington et al., 2006).

Multiple examples showing global and regional environmental change from the past and present naturally raise concerns about the future. A significant amount of effort from the scientific community in recent years has been devoted to projecting the future using computer models in order to provide the ability to bring together data and understanding of processes such that responses to drivers outside of recorded data can be simulated.

The use of models for impact assessments has been very widespread in hydrology over the past two decades (Borah and Bera 2004). However, few studies (e.g. Barron et al. (2012) in Australia, but only for climate change) have taken the analysis beyond the simulated effects on hydrological regimes (such as river flow or groundwater levels). Combining the use of a hydrological model with an indicator-based approach to defining environmental flow requirements would facilitate an extension of science to the indirect future effect on river flow-dependent biota. Such an approach, proposed in this paper, is a new contribution to the science of environmental flows. There has been a recent tendency to integrate water quality within environmental flows (Nilsson and Renöfält 2008) as quantity and quality are in many cases, strongly linked. However, in this study of environmental flows, we are focusing on the quantity of water, through time, required to maintain river health in a particular state (Acreman and Dunbar 2004). Whilst it is generally accepted, for example, that land use intensification can cause deterioration in water quality, which ultimately affects aquatic biota (Norris et al. 2007), this requires a different modelling approach and a different set of metrics and critical values. In our study here we concentrate on the less obvious, and hence more challenging, definition of ecologically relevant flow alterations that will occur under intensified land use. The results of this study are presented in a spatially-explicit manner, providing qualitative indicators of the extent to which river flow-dependent biota, characteristic of semi-natural lowland rivers in Central European plains, might be affected by land use and climate change-driven impacts in 2050s.

The general aim of this paper is to assess the impact of four scenarios that combine changes in climate, atmospheric CO_2, land and water use possible by 2050 on the specific set of ecologically-relevant flow regime indicators that define environmental flow requirements. This aim is presented through a modelling case study of a semi-natural river basin in Poland.

2 MATERIALS AND METHODS

2.1 Study area

The River Narew is situated in northeast Poland (Fig. 1) and its basin area upstream of the Zambski Kościelne gauging station is ca. 28,000 km^2. The characteristic features of rivers in this lowland area are their low slopes and large floodplains, that have good connectivity with main channels due to lack of embankments. Mean January and July temperatures are -3 and 17°C, respectively, while annual mean precipitation is approximately 600 mm. Flood peaks are associated with snow melt, which usually occurs in early spring or during warmer spells of winter. The magnitude of flooding can vary to a large extent between dry and wet years. The period between July and September is typically the low-flow period.

Sands, loamy sands, sandy loams and organic soils are the dominant soil types, while the dominant land use is agriculture (46% as arable land and 17% as grassland). The forests occupy about one-third of the area. Population density is low (ca. 59 pers./km^2, two times less than Poland's average) and there is nearly no heavy industry. Extensive use of land by farmers predominates. Many of river valley bottoms are in a virtually natural state and are protected as national parks or Natura 2000 sites. Flows in the Narew River basin are not significantly affected by regulating impoundments (weirs and dams) or water abstractions and discharges (Piniewski *et al*. 2011), especially when compared with the whole area of Poland or Western European countries. A more detailed description of the physiographic and socio-economic aspects of this region can be found in Piniewski (2012).

2.2 Hydrological model

Because future river flows have, by definition, not been measured, fulfilment of the objective of this paper requires a means of simulating future flow time series. Specification of future flows is achieved by using current data to understand the processes by which precipitation generates flow (given other factors, such as temperature) and then driving these relationships with projections of precipitation and temperature under future climates. Computer models provide an ideal tool for this challenge. There are many models available, but for this application we need a model that incorporates land use so that we can also simulate the implications of land use change. Lumped conceptual models (Post and Jakeman 1999) simulate flow at fixed points in a river system and use very simplified representation of rainfall-runoff processes. We have selected to use a distributed, physically-based, catchment-scale hydrological model as this is more explicit in its representation of processes than lumped models and produces results at points throughout a catchment. Distributed models vary with respect to discretisation strategy: from fully-distributed, grid-element based models, such as MIKE SHE (Refsgaard and Storm 1995), to semi-distributed models built on the concept of hydrological similarity, such as TOPMODEL (Beven and Freer 2001) or the Soil and Water Assessment Tool (SWAT; Arnold *et al*. 1998). The latter was selected as the modelling tool in this study.

SWAT model

SWAT is a river basin-scale model developed to quantify the impacts of land management practices in large, complex river basins (Arnold *et al*. 1998). SWAT is a continuous time model that operates on a daily time step and simulates the movement of water, sediment and nutrients on a catchment scale. The river basin can be partitioned into a desired number of sub-basins based on the Digital Elevation Model (DEM). The smallest unit of discretisation is a unique combination of land use, soil and slope overlay, referred to as a “hydrological response unit” (HRU). Runoff is predicted separately for each HRU, and then aggregated to the sub-basin level and routed through the stream network to the main outlet, in order to obtain the total runoff for the river basin. Key processes associated with the land and routing phase of the hydrological cycle included in SWAT were described in the theoretical documentation (Neitsch *et al*. 2011). In this study SWAT is applied solely to simulate river flows, hence sediment and nutrient movement simulation by SWAT is outside the scope of this paper. SWAT2009 (revision 481) model version (Neitsch *et al*. 2011) was used in this study.

SWAT set-up, calibration and validation

The preliminary set-up of SWAT for the NRB was established by Piniewski and Okruszko (2011) and was then substantially developed by Piniewski (2012). The latter set-up is applied in this paper. A brief description follows below, while the reader is referred to the afore-mentioned publications for more details.

The NRB was divided into 151 sub-basins and 1131 HRUs. Land use codes had to be reclassified from the CORINE Land Cover 2000 land use classification into the classification used in the SWAT Land Cover/Plant Growth database. Eight different classes were distinguished, of which three major classes were arable land (46.3%), evergreen forests (23.2%) and grasslands (17.3%). Twenty seven soil classes were distinguished based on the map of the benchmark soil profiles provided by the Institute of Soil Science and Plant Cultivation, in Puławy. Sandy loams, sands and loamy sands represented the three dominating classes, occupying 26.7%, 25.3% and 21.1% of the total basin area, respectively, while the percentage of peat soils was considerably high as well (16.9%).

Climate data were interpolated outside the model from gauge locations to SWAT sub-basins using the Thiessen polygon tool in ArcGIS. The original gauge data included 78 precipitation gauges and 14 climate (i.e. air temperature, relative humidity and wind speed) gauges, with daily data covering the time period from 1986 to 2008.

Daily mean river flow data from 27 flow gauges covering the period 1989-2008 were used for calibration and validation. Using such a large number of stations has been rare in calibration of models like SWAT, since more often single-gauge calibrations are undertaken. Spatial (multi-site) calibration allows for a comprehensive assessment of model performance at various spatial scales. Automatic calibration software provides tools for assessment of parameter sensitivities and performing spatial calibration in a systematic way. SWAT Calibration and Uncertainty Program (SWAT-CUP; Abbaspour 2008) was applied for model calibration and the Particle Swarm Optimisation (PSO) tool included in this software was selected as the optimisation method. Nash-Sutcliffe Efficiency was employed as the objective function. Calibration period covered years 2004-2008, whereas (temporal) validation period 1989-2003. Apart from temporal validation, spatial validation was performed using data from 12 stations that were not used in calibration. The full description of the calibration and validation strategy as well as calibration parameters can be found in (Piniewski 2012).

Figure 2 illustrates the results of calibration and validation of SWAT in the NRB. The median values across all 27 gauges (0.58 and 0.50 during calibration and validation periods, respectively) indicate acceptable model performance in both periods. Indeed, as suggested by Moriasi *et al*. (2007), approximate threshold for a satisfactory model performance in the case of monthly flow is NSE > 0.5, while in the case of daily flow, it is justifiable to use less stringent threshold, e.g. 0.4. In the calibration period seven out of 27 gauges had NSE values smaller than 0.4, but all of them had NSE larger than 0. In calibration period all 27 gauges had their absolute values of percent bias (PBIAS) smaller than 25%, while in validation period this threshold was exceeded in four cases. When cross-comparing goodness-of-fit measures between different phases of spatial calibration and validation, it is evident that, in general, the farther downstream, the higher NSE. The potential reasons for such model behaviour were discussed in Piniewski (2012). Hence, a correlation of NSE with the logarithm of the catchment area is present (Fig. 2B and 2C), with a coefficient of determination equal to 0.72 and 0.66 , respectively, for the calibration and validation period, respectively. This observation allowed us to focus our attention in further studies to selected sub-set of reaches, excluding reaches upstream of the gauges for which the model performance was not satisfactory. Nine gauges (all with upstream catchment areas below 1,000 km^2) were thus rejected and the spatial extent of analysis was limited to 52 reaches that remained. In principle, all major rivers of the NRB were kept, while all small tributaries and upper parts of medium tributaries were excluded.

Figure 3 shows simulated and observed daily hydrographs in the calibration period and daily flow duration curves (FDCs) for the combined calibration and validation period for four selected stations, one per sub-region. In general, visual inspection confirms previously mentioned statistical evaluation. Simulated discharge of the R. Narew at Suraż and of the R. Biebrza at Osowiec has generally lower variability than the observed discharge (Fig. 3A-D). In particular, some of the flood peaks are under-estimated and some of the low flow periods are over-estimated. The low flow tail of the FDC for Osowiec shows that for low exceedance probabilities simulated discharges are systematically over-estimated (e.g. Q95 is over-estimated by 34%). The R. Pisa (Fig. 3E-F) has a remarkably different flow regime than the two previously-mentioned rivers due to the occurrence of lakes in its drainage area. Given that SWAT does simulate hydrological effects of lakes in a simplified manner, the results for the R. Pisa are satisfactory. The best fit of simulated to observed values can be observed for the main outlet, the R. Narew at Zambski Kościelne (Fig. 3G-H).

2.3 Future scenarios

Two different types of future changes in the NRB were distinguished: those that are the consequence of global changes (climate and CO_2 change) and those that are specific to the NRB (land and water use change). The impacts on the hydrological cycle of climate and CO_2 change were represented in the model in a standard way, using downscaled projection from two GCMs. Land and water use change were represented in SWAT following the results of the complex scenario development process carried out in the NRB in 2008-2011 within the European Commission FP6 SCENES project (Kämäri *et al*. 2008; Giełczewski *et al*. 2011).

Climate and CO_2 change scenarios

The climate change signal for the time period 2040-2069 (hereafter 2050s) was derived from the output of two different GCMs: IPSL-CM4 from the Institute Pierre Simon Laplace, France (Marti *et al*., 2006), and MIROC3.2 from the Center for Climate System Research, University of Tokyo, Japan (Hasumi and Emori, 2004), both forced by the SRES-A2 emission scenario (IPCC, 2007). Downscaled precipitation and temperature projections from these two GCMs were used in the SCENES project to drive the continental-scale hydrological model WaterGAP (Schneider *et al*. 2011). The delta-change approach was used to reduce GCM biases. Based on the assumption that GCMs more accurately simulate relative change than absolute values, a constant bias through time is assumed in this approach. The delta-change factors (DCFs) are calculated at the monthly time-scale and spatial scale of SWAT sub-basins, using the future and present downscaled GCM outputs. For temperature, DCFs are defined as arithmetic differences between the future and present long-term means, whereas for precipitation, which is a multiplicative variable, future to present long-term mean ratios are defined. Table 1 shows basin-averaged monthly DCFs for temperature and precipitation under two selected GCMs for the 2050s. Both climate models project similar increases in mean annual temperature, although the seasonal variability of this increase is slightly different. The GCM projections of precipitation change show much higher levels of uncertainty than the projections of temperature change. In general, MIROC3.2 projects more variability than IPSL-CM4, according to which relative changes in precipitation do not exceed +/-25% for any month and mean annual precipitation is almost the same as in the baseline. According to MIROC3.2, there is a projected 11% increase in annual precipitation and only in July is the sign of change negative.

In addition to modification of the climate signal, in the current study we simulate hydrological effects of elevated atmospheric carbon dioxide (CO_2), which is a fundamental element in emission scenarios driving the GCMs. Elevated CO_2 is reported to alter plant growth through decreasing stomatal conductance and increasing leaf area index (LAI). The former change decreases evapotranspiration (ET), whereas the latter change has an opposite effect. According to several studies (Li *et al*. 2010, Warren *et al*. 2011), the combined effect leads to a decrease in ET.

CO_2 concentration is a static parameter in SWAT and is set to 330 ppmv. This default value was used in the baseline simulation. In the future scenarios this value was increased by 48% according to the assumptions of SRES-A2. We followed the suggestions of Wu *et al*. (2012) who applied land cover-specific modifications of two SWAT parameters: maximum stomatal conductance (GSI) and maximum leaf area index (BLAI). The details can be found in Piniewski (2012). It is believed that changes in plant physiological parameters provide a more physically justified representation of future impacts of elevated CO_2 in the model.

Land and water use change scenarios

The SCENES project produced a set of water scenarios for pan-European freshwaters up to 2050s, developed using novel scenario development techniques, at a range of scales (Kämäri *et al*. 2008). The NRB was one of the pilot areas in SCENES in which a series of stakeholder workshops was held as a part of scenario development process (Giełczewski *et al*. 2011). These workshops developed two scenarios: Sustainability Eventually (SuE) and Economy First (EcF). The SuE storyline was perceived by the stakeholders as an environmentally optimistic, plausible and desired future (and as a continuation of currently observed trends, showing that the NRB has developed in a sustainable way so far). In contrast, the message coming out of the EcF storyline was rather negative, particularly for the environment. However, this scenario produced a faster economic growth for this region, mainly through a more intensive agriculture bringing much higher crop yields (Giełczewski *et al*. 2012). It was also emphasised by stakeholders that this scenario is not very likely to happen in the NRB (yet, it is plausible) and its realisation in the future would require a push by an external factor.

Piniewski (2012) converted qualitative scenarios of the NRB development created within SCENES into model representations using a 3-step conversion protocol developed by Alcamo (2008), being a part of the Story-And-Simulation (SAS) method. Scenarios were not uniform across the whole NRB, but were specific for four sub-regions: Upper Narew, Biebrza, Great Masurian Lakes (GM Lakes) and Lower Narew (LNB), as shown in Figure 1. In this study we are using the same two converted model scenarios as Piniewski (2012). A brief description of the main driving forces behind these scenarios as well as quantitative changes in parameters follows below.

Three types of land use change were considered in scenarios: (1) between agricultural land and forests, (2) between built-up areas and agricultural land and (3) within agricultural land, the change between arable land and grasslands. The first describes afforestation or deforestation processes, the second urban growth, while the third refers to the broad direction of agricultural development. In addition to changes between land use types, also the actual use of agricultural land was considered in scenarios. At present, the NRB is characterised by extensive agriculture with low fertilisation rates. The scenarios include assessment of whether agricultural production will intensify in the future or not. This was achieved through consideration of changes in mineral and organic fertiliser amounts (parameter FRT_KG) in agricultural HRUs. Changes in fertilisation rates have a direct impact primarily on water quality, while an indirect impact on water quantity could theoretically be reflected in altered water uptake by crops, which is caused by altered crop yields under different fertilisation rates (Rose *et al*., 2012).

As mentioned in section 2.1, at present the NRB has a low population density and virtually no heavy industry. Local stakeholders generally agreed that this would not change in the future (Giełczewski *et al*. 2011). Hence, no future changes in water abstractions for households and industry are included. In contrast, changes in irrigated area were included, as there is a high likelihood of an increase in irrigated area in the NRB because changes in the form of ownership in agriculture in the early 1990s led to the abandonment of a large number of irrigation systems (Łabędzki 2007). In the NRB irrigation occurs almost solely as drainage sub-irrigation systems in soils with shallow groundwater depths cultivated as grasslands. In SWAT, irrigation is represented by scheduling auto-irrigation operations in selected HRUs. Thus, if a percentage of irrigated grassland is going to change under a given scenario, this can be reflected in SWAT by adding or deleting an auto-irrigation operation to/from a certain number of HRUs. In addition, future changes in arable land areas equipped with tile drainage were also considered.

The computed trends in driving forces composing two analysed scenarios for four sub-regions of the NRB are shown in Figure 4. They were translated into modified SWAT parameter values only for the future period of 2050s, so that they conform to climate change scenarios. The principal land use change in the SuE scenario was from arable land to forests and grasslands, while the opposite change took place in the EcF scenario. Fertilisation rates generally decrease (mineral fertilisers) or do not change (organic fertilisers) under SuE, while under EcF they increase substantially, which leads to higher crop yields and higher nutrient losses (Giełczewski *et al*. 2012). In both scenarios a small or medium increase in irrigated areas and areas equipped with tile drainage is expected by 2050. Variability in trends between sub-regions was rather small, especially for the EcF scenario.

2.4 Environmental flow requirements

The building block method embraces the fact that all components of the flow regime, including low flows, high flows, freshets, etc., have ecological significance and an environmental flow regime can be constructed by combining together the element of the flow hydrograph required to deliver ecological objectives. Blocks may be defined for specific species, biological communities or to maintain underlying processes such as sediment transport (which maintains habitat and morphological structure) or river-floodplain connectivity. Figure 5 shows a generic example of the building blocks of the flow regime derived for UK rivers (Acreman *et al*. 2009). The literature review undertaken by Acreman *et al*. (2009) demonstrated that the same species often had variable flow requirements from site to site, and further suggested that locally available information and expertise should be used as much as possible to define these building blocks. The initial step in recognising ecological flow requirements typical for the NRB's species was carried out by Piniewski *et al*. (2011).

The environmental flow regime of the NRB rivers consists of three blocks, one dealing with low flows and two dealing with floods (Fig. 6). The first block was defined using a method well-established in Poland, known as the Kostrzewa method. This block satisfies the basic requirements of aquatic fauna with respect to minimum in-stream flow requirements (Kostrzewa 1977). In the Kostrzewa method, the minimum in-stream flow threshold is defined as a function of mean annual minimum flows, catchment area and geographical location. This approach corresponds to the look-up table methods of environmental flow assessments (Acreman and Dunbar 2004) and is routinely used in Poland to design hands-off flows for managing water systems and abstraction licences.

The second and third blocks were defined using the novel approach developed by Piniewski *et al*. (2011) and extended in this study. This approach builds upon the concepts of umbrella and flagship species existing in conservation biology (Simberloff 1998). The second block provides spawning and nursery habitats for pike, which is a key fish species in semi-natural lowland rivers in Poland and a good indicator of ecological health of a river (cf. Penczak and Koszalińska 1993; Piniewski, 2012). It is generally accepted that pike spawning success can be enhanced by flooding due to a rapid increase in its preferred marshy spawning ground area (Inskip 1982). Hence, this block is defined by the appropriate timing and duration of flooding (any consecutive 20 days with flows exceeding bankfull flow threshold between 1 March and 31 May). Similarly, Denic and Geist (2010) identified habitat suitability of lacustrine brown trout as a flagship species for the river-lake system in Bavaria, Germany.

The third block maintains floodplain vegetation communities in good health. It is also defined using the timing and duration of flooding as key variables, whereas duration is not a fixed value for the whole basin, but may vary depending on dominant plant communities along the reach (Table 2; cf. Piniewski 2012). A similar, though more sophisticated approach of using a group of flagship wetland plant (among other) species for identification of inundation requirements of large floodplain wetlands was applied by Rogers *et al*. (2012) in the Murray-Darling Basin, Australia.

The inherent danger is that although direct relationships between flow characteristics (e.g. flood) and ecosystem response may be in some cases summarised by simple rules, the final constructed flow regime may lack crucial characteristics that support other ecosystem features in complex indirect ways, such as through controlling food web interactions (Shenton et al., 2010).

2.5 Development of indicators

In order to assess to what extent the water requirements of biota are met in the control period and in the future scenarios, an approach common in water resource systems was utilised, whereby water demand of river-dependant ecosystems is characterised by the mathematical representation of the building blocks and related to available water supply modelled at river reach scale with a daily time step using SWAT. In this approach, the combination of water supply and water demand leads to the calculation of indicators that evaluate the performance of a water resource system (Hashimoto *et al*. 1982). Kundzewicz and Kindler (1995) suggested that the system performance is a binary variable and can be satisfactory (S), if water demand is less than water supply, or exceeds water supply (NS) at a given time. In this study we will adapt for our purposes those of the reliability measures used in water resource systems that refer to time period that a system spent in state NS.

For the first building block (minimum in-stream flows) state NS is characterized by a modelled flow for a given day being below the Kostrzewa threshold. A corresponding temporal reliability indicator MIF is thus defined as the ratio of time the system is in state S to the total time period considered (here 20 years).

For the second building block (floodplain inundation for pike spawning) state NS is first evaluated on annual basis. For a given year i, let t_i denote the maximum duration of floodplain flooding between March and May. State of the system is characterized by the annual reliability R_i quantified by comparing t_i with the optimal duration of floodplain flooding for pike D_{opt} :

$$R_i = \min\left\{1, \frac{t_i}{D_{opt}}\right\}$$

where $i = 1,2,\ldots,20$ and D_{opt} is equal to 20 days. R_i expresses relative frequency of demand being satisfied in a particular year and R_i equal to 0 implies state NS, while R_i equal to 0 implies state S. Mean annual reliability indicator $PIKE$ is then calculated as an arithmetic mean of R_i.

For the third building block (floodplain inundation for vegetation communities) state NS is also first evaluated on annual basis. For a given year i, let τ_i

denote the duration of floodplain flooding between March and October (hence, $\tau_i \in \{1,2,\ldots,245\}$). As shown in Table 2, each river reach distinguished in SWAT was assigned one of four categories of optimal/critical durations of floodplain flooding depending on dominant vegetation community. State of the system is characterized by the annual reliability S_i quantified by comparing τ_i with the characteristic durations of floodplain flooding:

$$S_i = \max\left\{\min\left\{\frac{\tau_i - a}{b - a}, 1, \frac{d - \tau_i}{d - c}\right\}, 0\right\}$$

where $i = 1,2,\ldots,20$ and S_i denotes the trapezoidal curve defined by parameters a, b, c and d (cf. Tab. 2). The parameters a and d locate the "feet" of the trapezoid and the parameters b and c locate the "shoulders". S_i expresses relative frequency of demand being satisfied in a particular year and S_i equal to 0 implies state NS, while S_i equal to 0 implies state S. It is noteworthy that in contrast to pike, floodplain vegetation communities can be in state NS also if there is "too much" water (i.e. $\tau_i > d$). Mean annual reliability indicator *FVC* is then calculated as an arithmetic mean of S_i.

A central feature of this research was to analyse how these indicators would change in the future. The change is measured as the absolute difference ΔX between an indicator X calculated for a given model scenario X_m and for the baseline period X_b. If $\Delta X < 0$, then conditions are supposed to worsen, while if $\Delta X > 0$ the opposite takes place. In order to present the results in a manner readily understandable by water managers and decision-makers, a consistent colour-coding system composed of seven classes of impacts was developed. The system includes gradual changes in both directions: three classes referring to a positive change (small, moderate and large, marked in different tones of green), three referring to a negative change (small, moderate and large, marked in different tones of red) and one class referring to insignificant change (marked in grey). Developing a colour coding system requires the definition of thresholds, which is often quite problematic. If, as in this case, no indicator values are reported in the literature, thresholds have to be determined based on expert judgement. Threshold-based traffic light-type colour coding systems were developed to map Europe-wide environmental flow indicators in the SCENES project (Piniewski *et al*. 2013; Okruszko *et al*. 2012, Laizé *et al*. 2013). The threshold values from Table 3 were determined using standard deviations ΔX as a natural measure of variability. For each indicator the "small change" class was first defined in such a way that it would contain the standard deviation. The remaining thresholds defining other classes were then set proportionally and symmetrically.

2.6 Experimental design

In total, five model runs were conducted. The baseline scenario refers to the calibrated model run driven by the observed climate data for the control period of 1989-2008. Four scenario runs, referring to the future period of 2050s, are all possible combinations of GCMs with land and water use scenarios. Hence, indicators of change between future periods and baseline situation were computed and indicator maps were created for each of the combined scenarios. The latter were named: *IPSL_SuE*, *IPSL_EcF*, *MIROC_SuE* and *MIROC_EcF*, where for example *IPSL_SuE* refer to the combined scenario of climate change from IPSL-CM4 with land and water use change as in the scenario Sustainability Eventually (cf. section 2.2). Maps were created in a semi-distributed manner, showing variability at sub-regional level, and not at the reach-scale level. As mentioned in section 2.3 the NRB was divided into 4 sub-regions upon development of the land and water use scenarios and this division is used for analysing the results.

3 RESULTS

Figures 7-9 illustrate the projected impact of future alterations to river ecosystem elements through environmental flow indicators related to: minimum in-stream-flows (*MIF*, Fig. 7), flows for pike spawning (*PIKE*, Fig. 8) and flows for floodplain vegetation communities (*FVC*, Fig. 9). Pie charts show percentages for seven impact classes summarized in Table 3 in each sub-region (pie diameter is proportional to the number of reaches in a sub-region: 14, 10, 2 and 26 in Upper Narew, Biebrza, GM Lakes and Lower Narew, respectively).

The indicator maps in Figure 7 show changes in the ratio of time the modelled flows are above the minimum in-stream flow threshold to the total time period considered. An increase in the indicator value for a particular scenario refers to a positive situation, when the duration of time in which the system is in state NS is shorter than in the baseline period. Variability in this indicator can be observed mainly in terms of climate scenario, while variability in terms of land and water use scenario as well as sub-region is considerably smaller. Overall, 'Insignificant change' classes dominate under scenarios driven by MIROC3.2, while negative impacts dominate under those driven by IPSL-CM4. It is noteworthy that in 42 out of 52 reaches the system was in state S for 100% of the time during the baseline period, which means that current situation is so good that there is no room for improvement in the future; this explains the predomination of the 'Insignificant change' class under climate scenarios driven by MIROC3.2. In addition, in general more negative impacts can be observed under scenarios associated with SuE than those associated with EcF. 'Large negative' impact class was assessed only in the Biebrza sub-region.

Indicator maps in Figure 8 show changes in the number of years with sufficient water for pike spawning (as described in section 2.5). It is noteworthy that two scenarios driven by IPSL-CM4 can be characterised by negative changes, whereas two scenarios driven by MIROC3.2 are indicated by positive changes. In contrast, the differences between indicator maps associated with SuE and EcF for the same GCM are rather small. As with the previous indicator, projections of change are a little more optimistic for the market-driven future (EcF) than for the sustainable future (SuE). Sub-regional variability is less spectacular than inter-scenario variability, but it is also present in some of the maps. For example, under *IPSL_EcF*, more than half of the Biebrza reaches belong to the 'Insignificant change' class, whereas over 70% of the Upper Narew reaches to the 'Moderate' or 'Large decrease' classes.

Indicator maps in Figure 9 show similar patterns to those in Figure 8: they depict the number of years with the appropriate amount of water required for floodplain vegetation communities. However, the *FVC* indicator is additionally dependent upon the dominating floodplain vegetation community category and the resulting optimal duration of inundation (cf. Tab. 2), hence its interpretation is more difficult. The variability of impacts among sub-regions is considerably larger than in the two previous maps, which is directly connected to the spatial variability in the dominating plant communities. In particular, none of the previous indicators showed all seven possible classes of impacts within one sub-region, as now happens for *MIROC_EcF* in the Lower Narew. Thus, spatial variability in Figure 9 masks variability between different categories of plant communities. Table 4 provides additional valuable insight into the results by presenting basin-averaged impact classes for different categories of floodplain vegetation communities under four analysed scenarios. It can be observed that in all four cases projected impacts for 'No inundation' category have their sign opposite to the signs of impacts for other three categories. This is because for dry and mesic meadows (the dominant vegetation communities in this category) more frequent flooding does always imply worse conditions (cf. Tab. 2). Furthermore, in Table 4 the same as in Figures 7 and 8, the differences between the results obtained for SuE and EcF under a given climate change scenario is considerably smaller than the differences between the results obtained for IPSL-CM4 and MIROC3.2 under a given land and water use scenario. The most valuable plant communities, sedge and tall sedge meadows, belong to the categories of medium- and long-term inundation, respectively. It can be seen in Table 4 that they are vulnerable to small/moderate negative changes under the IPSL-CM4 climate and a small positive change under the MIROC3.2 climate.

It is worth noting that in the case of the two indicators associated with flooding (Figs. 8 and 9), the magnitude of impacts (either positive or negative) in the Upper Narew is always larger than in the Biebrza sub-region. This might be explained by spatial differences in driving forces between sub-regions. For example, more extreme changes in winter and spring precipitation are projected by both climate models for the Upper Narew than for the Biebrza (cf. Piniewski 2012). The amount of precipitation falling in winter and spring is critical for the occurrence, magnitude and duration of spring flooding. Additionally, the magnitude of future land use change is also more extreme in the Upper Narew than in the Biebrza, as shown in Figure 4.

Changes to the flow regime are caused by alterations to the water balance, which may eventually alter environmental flow indicators. Understanding changes in the catchment water balance can provide a meaningful insight into the understanding impacts on river ecosystems. Figure 10 illustrates the monthly distribution of basin-averaged water balance components (precipitation, actual evapotranspiration and runoff) in the baseline period and absolute changes in the future scenarios. Changes in precipitation are projected only under climate change scenarios (discussed in section 2.2, cf. Tab. 1), while under SuE and EcF precipitation does not change. Projected changes in actual ET (evapotranspiration) are fairly consistent between all four scenarios, especially in winter and spring (wet seasons in terms of runoff in Poland) when all projections suggest an increase. Actual ET is influenced by both climate, CO_2, land and water use change. As noted in section 2.2, all four scenarios were run using the assumption on CO_2 change by 48% by 2050s (related to the SRES-A2 scenario), which partly explains consistent response in actual ET. Land and water use change has visibly more impact on actual ET in July and August than in any other month. A decrease in actual ET in these months for all scenarios, surprising in the context of projected large temperature increase (Tab. 1), is probably a complex effect of changes in precipitation (decrease in July), and actual soil water storage (depleted after wet winter and spring).

Scenario consistency in runoff simulations is considerably smaller. Under both scenarios driven by MIROC3.2 an increase in runoff is projected throughout the whole year, with an exception of very small decreases in June and July. In contrast, under both scenarios driven by IPSL-CM4 a decrease in runoff is projected throughout the whole year apart from winter month. In each case, combinations of climate scenarios with EcF produce more runoff than corresponding combinations with SuE in each month apart from the period from June to August, when land and water use change scenarios have negligible effects on runoff.

Overall, Figure 10 explains well the impacts on environmental flows displayed in Figures 7-9: generally more optimistic future under wetter scenarios associated with MIROC3.2 and more pessimistic future under drier IPSL-CM4.

4 DISCUSSION

Several authors have investigated the combined hydrological effects of climate (and CO_2, if applicable) and land use change (Tong *et al*. 2012, Tu 2009, Choi 2008, Chang 2003; Park *et al*. 2011). The studies differed with respect to the applied models, approaches to developing model scenarios, hydrological aspects of impact assessment and geographical conditions. Unfortunately, none of the study areas in which the hydrological models were applied had a similar geographical setting to the NRB. The common finding in all the aforementioned studies was that the impact of land use change did not remarkably exceed the impact of climate change in any case, which strongly supports the findings in this study. Only Tong *et al*. (2012) concluded that the magnitude of impacts (in this case on mean annual runoff) was comparable between these two stressors; however, under the driest and the wettest climate change scenarios, the impacts were higher than under the land use change scenario. In the studies of Tu (2009), Choi (2008) and Chang (2003) (runoff regime), and Park *et al*. (2011) (all water balance components), climate change impacts remarkably exceeded land use change impacts.

Scenario decomposition carried out in Piniewski (2012) revealed that it is the land use change that is responsible for changes in catchment water balance, whereas the water use change had a small effect in the NRB. Assessing impact indicators for sustainable (SuE) and market-oriented (EcF) scenarios produces the un-anticipated conclusion that the future of environmental flows is brighter under the latter scenario rather than under the former. This should be regarded as an achievement of applying environmental models, in that they can be used to test hypotheses concerning how catchment system functions (Beven and Alcock 2012). This example shows that such hypotheses can sometimes give counter-intuitive answers: changes in driving forces considered sustainable by stakeholders lead to worsening the status of environmental flows. However, this can be interpreted as a trade-off, producing a “greener” environment in terms of larger percentage of forests and extensive grasslands, but at the cost of surface water resources and potentially aquatic ecosystems. Modelling studies on land use change effects in temperate climate are generally consistent with the findings in this paper. Heuvelmans *et al*. (2005) applied SWAT to examine effects of land use change on water balance of a small catchment in Belgium. They reported that deforestation to arable land (similar to EcF) resulted in an increase in runoff by 22%. In contrast, Kovár and Vaššová (2010) applied the WBCM model in a small catchment in Czech Republic, concluding that conversion of ca. 10% of arable land into grassland (similar to SuE) caused a decrease in growing period runoff. Thomas et al. (2011) reviewed possible mitigation measures to sustain minimum runoff during low flow periods, indicating that both deforestation and conversion of grassland to arable land are meaningful measures. Our study added an important contribution to the current knowledge, showing potential ecological effects of flow alterations caused by land use change to aquatic and riparian ecosystems.

The uncertainty of our assessment has not been precisely quantified, although it is currently well-known that the uncertainties related to climate impact modelling grow into an envelope, or a cascade, starting from the unknown future society and ending in unknown adaptation responses (Wilby and Dessai, 2010). Gosling et al. (2011) reported that the differences in projected changes of mean annual as well as high and low monthly runoff between the two types of hydrological model are generally relatively small in comparison to the range of projections across the seven GCMs. We have applied only two climate models, driven by only one emission scenario, while various GCM-SRES combinations are known to produce very uncertain signal, both at global level (IPCC, 2007) and in Poland, in particular for precipitation (Kundzewicz and Matczak 2012). Secondly, as pointed out by Teutschbein and Seibert (2008), the delta-change approach is not able to address changes in future climate variability (e.g. major events will change by the same amount as all other events), so future studies should take advantage of the now readily available global bias-corrected daily climate datasets (e.g. WATCH forcing data; Weedon et al., (2011)) or, in the case of using RCM output, explore more sophisticated bias-correction methods (Teutschbein and Seibert, 2008). Finally, hydrological modelling itself is subject to various uncertainties. Even though we quantified the SWAT model performance extensively (cf. Piniewski (2012) for more detail) and limited our analysis only to the sub-set of river reaches that produced more reliable output than others, there still is much room for improvement, in particular in model predictive capability at smaller spatial scales. In summary, as a result of various uncertainties, transferability of our projections to other catchments is speculative. Nevertheless, we expect that due to similarity of climate and physiographic features, the results are to some extent representative for lowland and lakeland catchments of the South-Eastern Baltic Sea Basin (the areas situated at the edge of North European and East European plain). For the land and water use change, though, the results are based on the catchment-specific, stakeholder-driven scenario assumptions, and therefore could be generalized only on condition that similar scenarios are produced for catchments of interest.

5 CONCLUSIONS

Different ecologically-relevant flow regime indicators that define environmental flow requirements show both positive and negative responses to future climate and catchment change, with a higher tendency for negative responses. In general, warm projections from IPSL-CM4 combined with sustainable land and water use projections from SuE scenario produce the most negative changes, while warm and wet projections from MIROC3.2 combined with market-driven projections from EcF scenario the most positive changes for river ecosystems. Climate change overshadows land and water use change in terms of projected flow alterations and implications for aquatic and riparian biota. An increase in winter runoff and an increase in winter and spring actual ET are the least uncertain responses of hydrology to climate change signal in the NRB, and perhaps, in temperate climate of the areas situated at the edge of North European and East European plain. The results show that sustainability has different dimensions, and in particular, does not mean the same for terrestrial and aquatic ecosystems: possessing “green” land use requires storing more water in the catchment, which reduces runoff, river flows and water available for riverine ecosystems. Although they may benefit from improved water quality from decreased arable land and reduced fertiliser use. The results of this study are useful to local land and water management authorities responsible for designing appropriate management alternatives and adaptation strategies, however their full usability needs to recognize the high level of uncertainty inherent in this kind of modelling.

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Table 1. Basin-averaged intra-annual variability in mean temperature and precipitation for the baseline period and delta change factors (DCFs) under two GCMs for the 2050s.

Month	1	2	3	4	5	6	7	8	9	10	11	12	Mean
Baseline temp. (°C)	-2.0	-1.4	1.9	7.8	12.7	15.8	18.2	17.5	12.7	7.8	2.2	-1.6	7.6
DCF[1] IPSL-CM4 (°C)	3.6	3.1	3.4	3.7	3.6	3.1	3.0	3.2	3.7	2.7	4.7	3.6	3.5
DCF MIROC3.2 (°C)	4.2	4.0	3.7	2.4	2.8	3.2	3.0	3.5	2.9	2.8	2.5	3.5	3.2
Baseline prec. (mm)	33.8	32.1	35.8	38.0	53.1	66.0	72.8	67.7	58.5	47.0	43.1	38.9	48.9
DCF IPSL-CM4 (%)	17	3	4	-25	0	10	-6	0	2	-18	8	17	1
DCF MIROC3.2 (%)	14	1	28	8	1	4	-15	41	21	17	10	4	11

[1] DCF – monthly delta change factors (additive for temperature, multiplicative for precipitation)

Table 2. Categories of floodplain vegetation communities (FVC) determined with respect to the optimal duration of inundation and characteristic values defining trapezoidal membership functions.

Symbol	Dominant vegetation communities	Inundation category	Optimal duration (days)	Sub-optimal duration (days)	Critical duration (days)	a^1	b^1	c^1	d^1
FVC1	Tall sedge meadows	Long-term	>120	60-120	0-60	60	120	∞	∞
FVC2	Sedge meadows	Medium-term	60-120	30-60 or 120-180	<30 or >180	30	60	120	180
FVC3	Wet meadows and riparian forests	Short-term	15-60	0-15 or 60-120	0 or >120	0	15	60	120
FVC4	Dry and mesic meadows	No inundation	0	0-15	>15	-∞	-∞	0	15

[1] The values of a, b, c and d denote the characteristic points of the trapezoidal curve defining optimal./critical conditions for FVCs (cf. section 2.5).

Table 3. Colour coding system developed for mapping indicators.

ΔMIF	$\Delta PIKE$	ΔFVC	Impact type	Colour code
$[-1, -0.05)$	$[-1, -0.25)$	$[-1, -0.25)$	Large negative	
$[-0.05, -0.03)$	$[-0.25, -0.15)$	$[-0.25, -0.15)$	Moderate negative	
$[-0.03, -0.01)$	$[-0.15, -0.05)$	$[-0.15, -0.05)$	Small negative	
$[-0.01, 0.01]$	$[-0.05, 0.05]$	$[-0.05, 0.05]$	Insignificant	
$(0.01, 0.03]$	$(0.05, 0.15]$	$(0.05, 0.15]$	Small positive	
$(0.03, 0.05]$	$(0.15, 0.25]$	$(0.15, 0.25]$	Moderate positive	
$(0.05, 1]$	$(0.25, 1]$	$(0.25, 1]$	Large positive	

Table 4. Basin-averaged impact classes for different categories of floodplain vegetation communities under four analysed scenarios.

Inundation category	Number of reaches	*IPSL_SuE*[1]	*IPSL_EcF*	*MIROC_SuE*	*MIROC_EcF*
Long-term	3				
Medium-term	15				
Short-term	18				
No inundation	16				

[1] All colour codes were defined in Table 3.

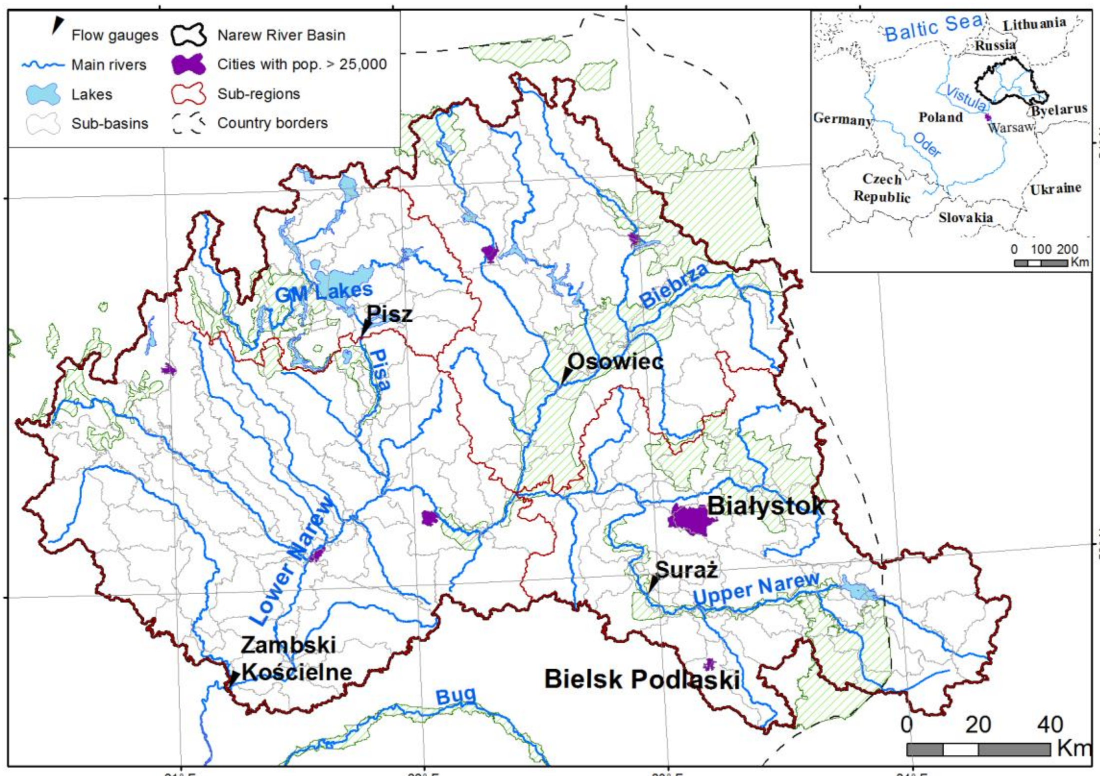

Fig. 1 Study area.

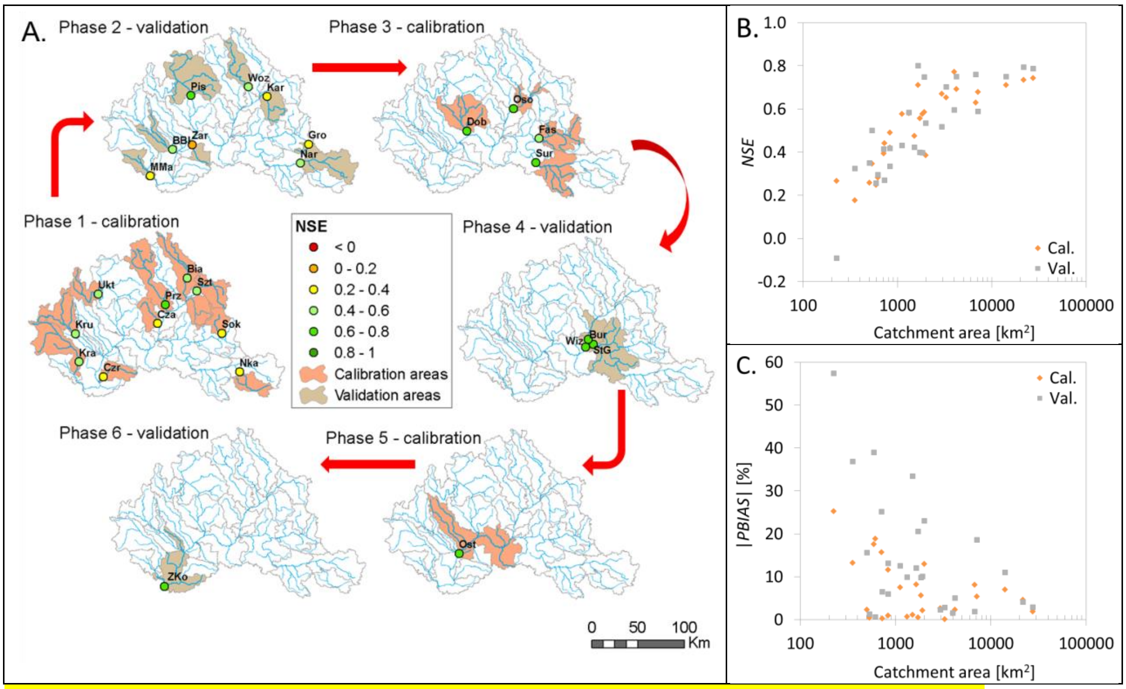

Fig. 2. Calibration and validation results: A. Iterative procedure of multi-site calibration and validation and spatial distribution of NSE; B. Relationship between NSE and catchment area upstream of the gauge during calibration period and validation period; C. Relationship between the absolute values of PBIAS and catchment area upstream of the gauge during calibration period and validation period.

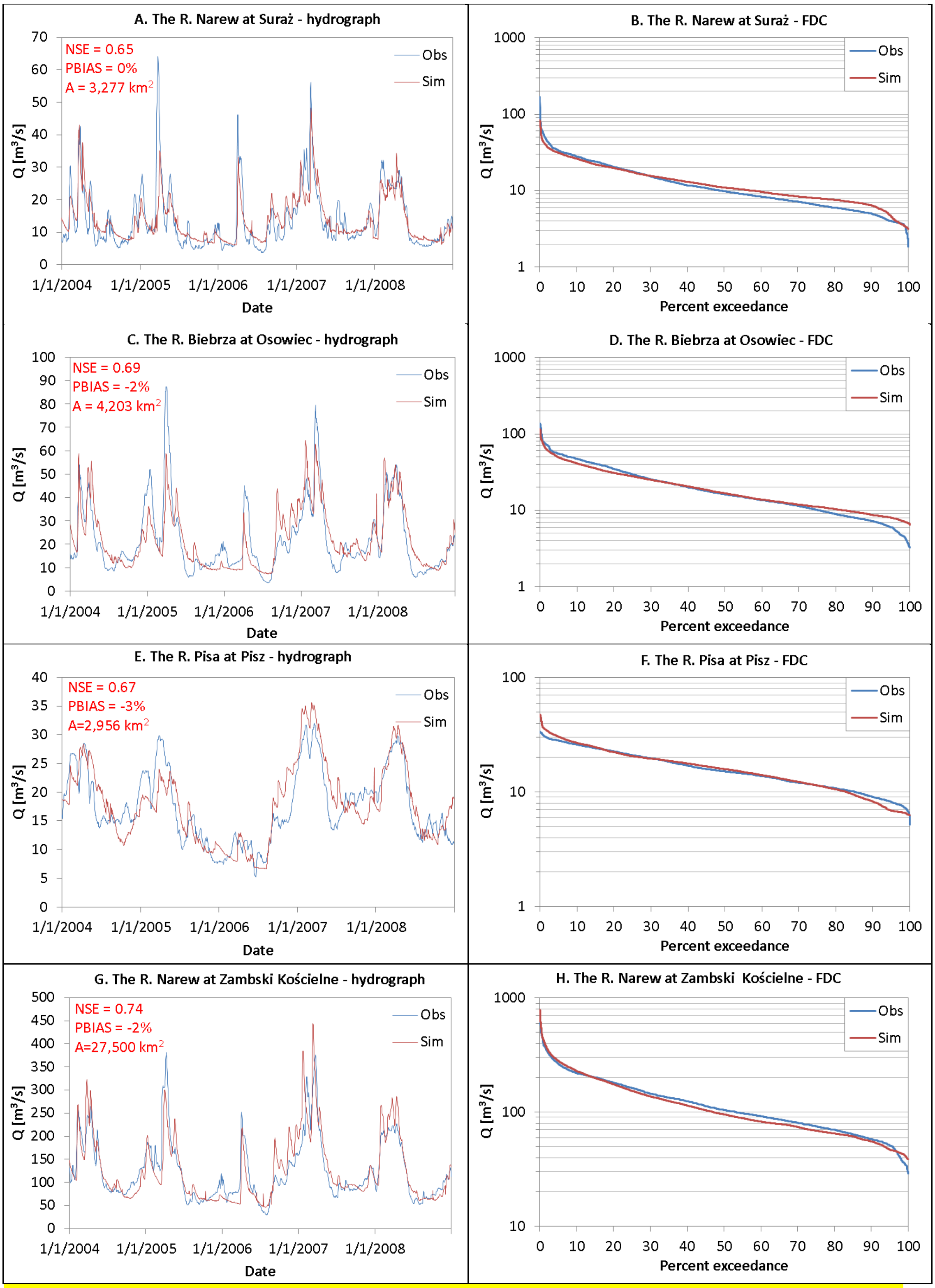

Fig. 3. Simulated and observed mean daily flows during the calibration period (2004-2008) and daily flow duration curves (FDCs) the combined calibration and validation period (1989-2008) for: the R. Narew at Suraż (A-B), the R. Biebrza at Osowiec (C-D), the R. Pisa at Pisz (E-F) and the R. Narew at Zambski Kościelne (G-H) (cf. Fig. 1 for locations of rivers and gauging stations). NSE – Nash-Sutcliffe Efficiency; PBIAS – percent bias; A – catchment's drainage area.

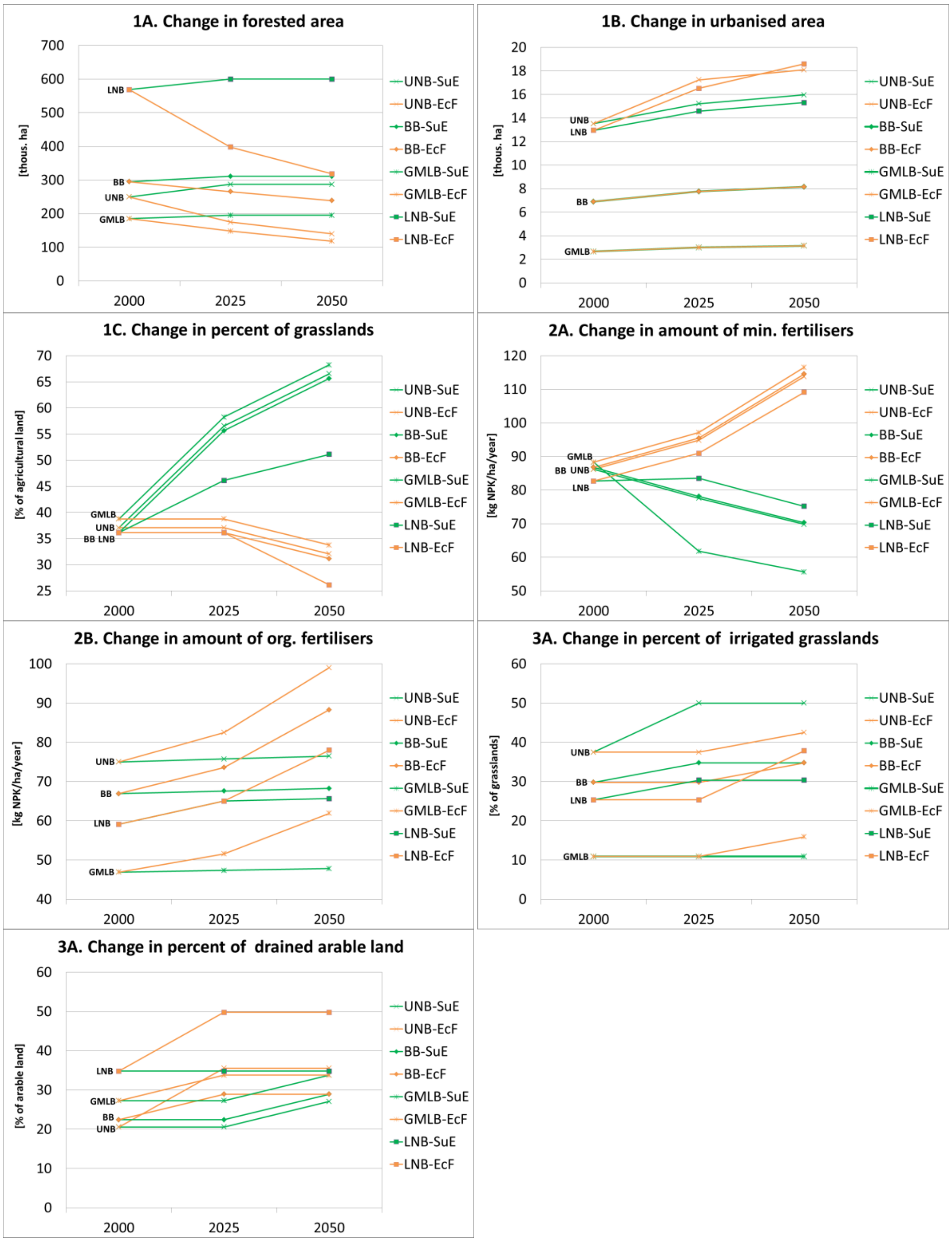

Fig. 4. Computed trends in driving forces composing two analysed scenarios for four sub-regions of the NRB (UNB - Upper Narew Basin, BB - Biebrza Basin; GMLB - Great Masurian Lakes Basin; LNB - Lower Narew Basin, cf. Fig.1).

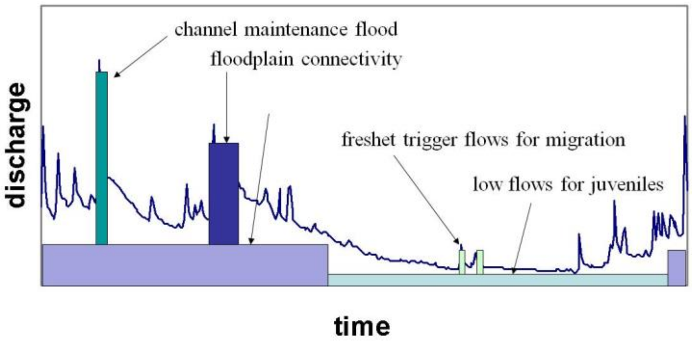

Fig. 5. Generic building blocks for environmental flow assessment (after Acreman *et al*. 2009).

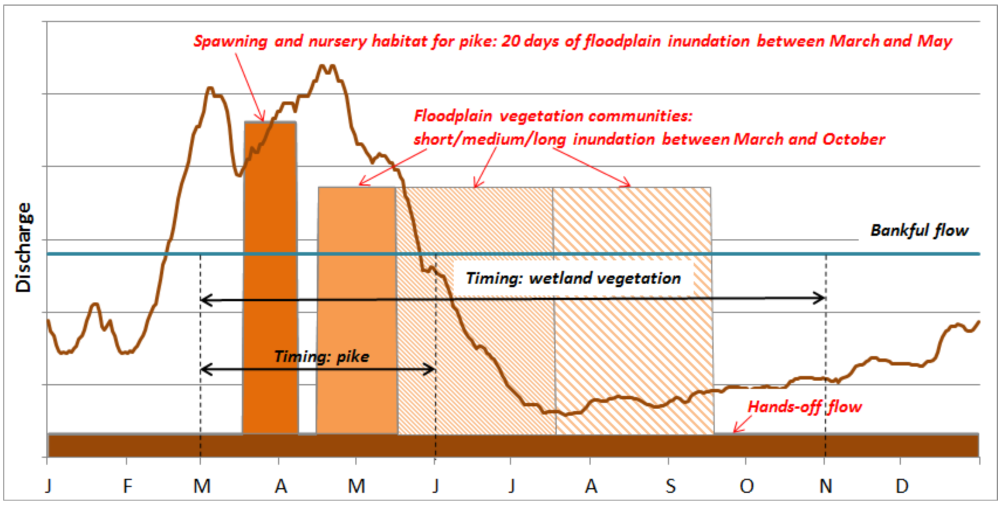

Fig. 6. Building blocks of the flow regime specific to the NRB.

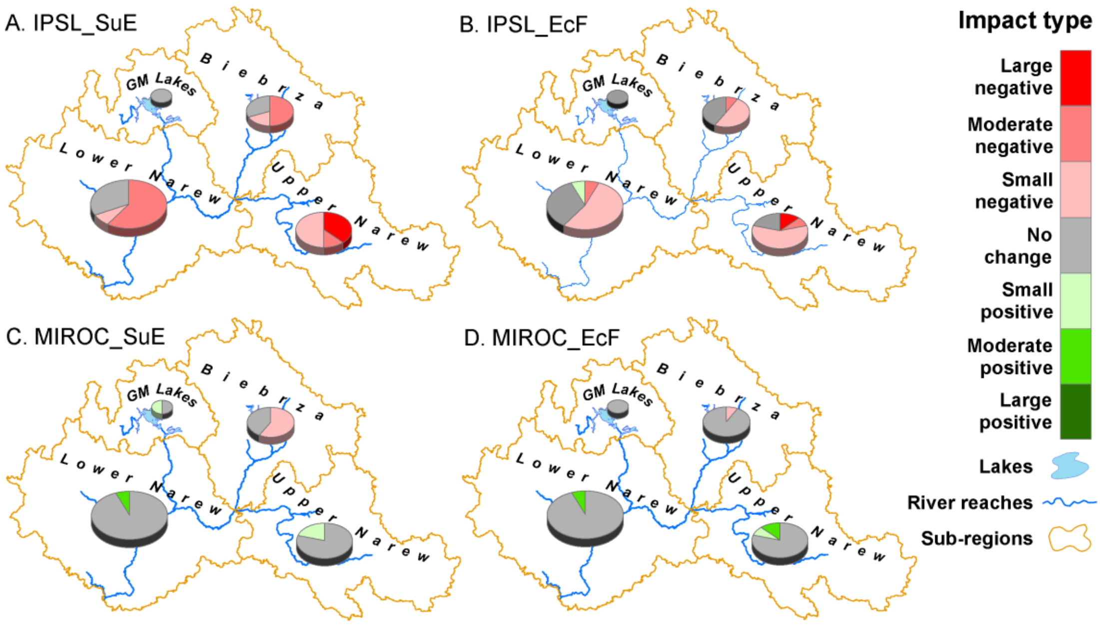

Fig. 7. Pie charts of different classes of impacts corresponding to the temporal reliability indicator *MIF* (minimum in-stream flows) under four analysed scenarios in NRB sub-regions (pie diameter is proportional to the number of reaches in a sub-region).

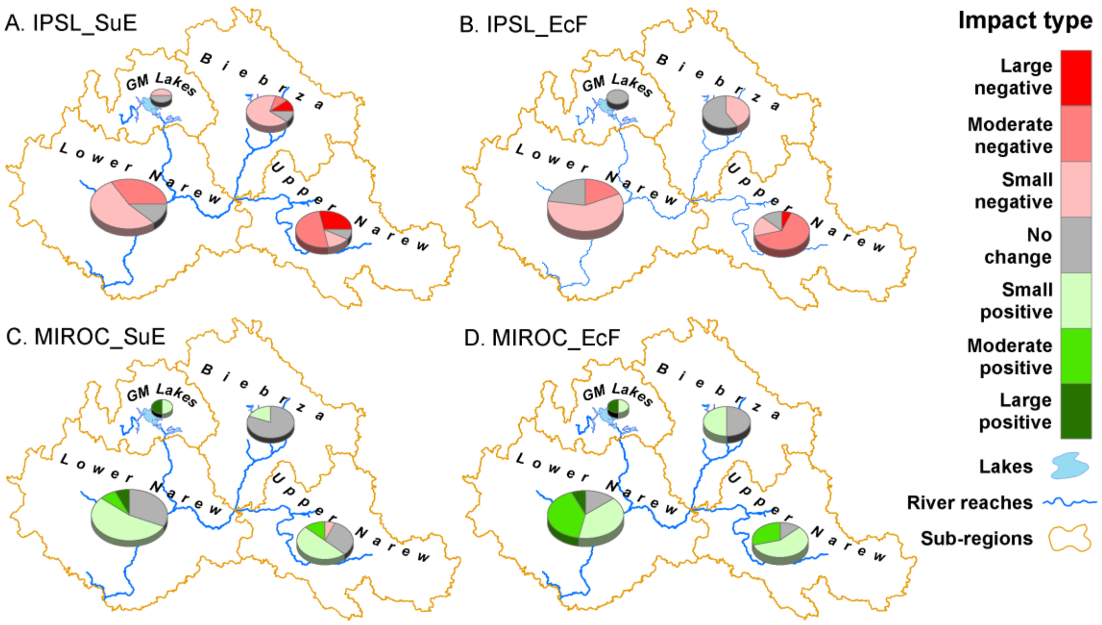

Fig. 8. Pie charts of different classes of impacts corresponding to the mean annual reliability indicator *PIKE* (provision of environmental flow for pike spawning) under four analysed scenarios in NRB sub-regions (pie diameter is proportional to the number of reaches in a sub-region).

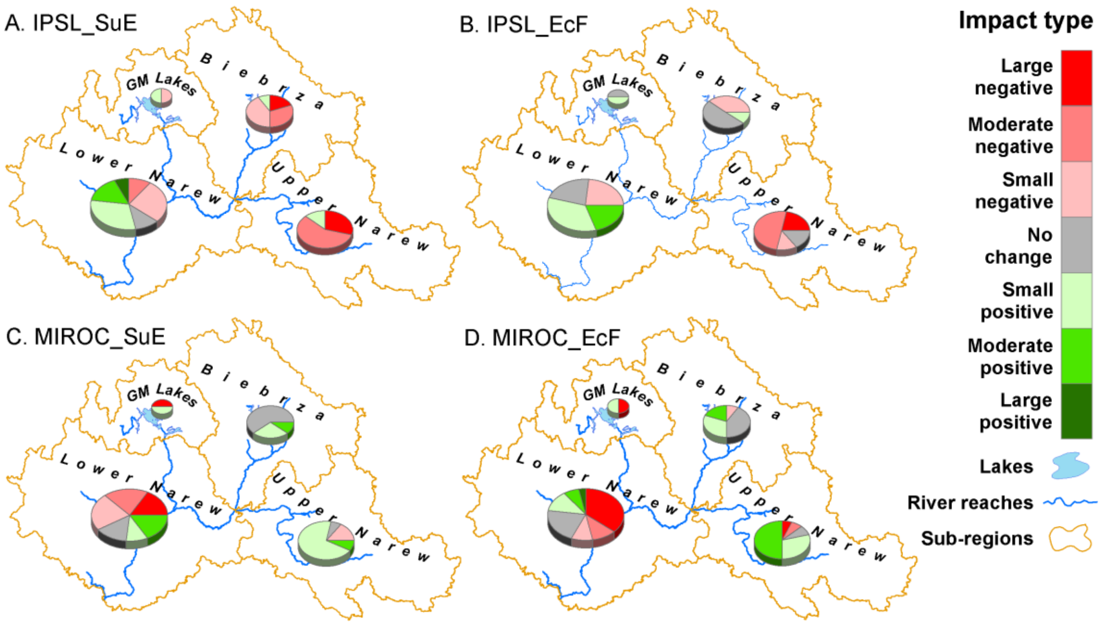

Fig. 9. Pie charts of different classes of impacts corresponding to the mean annual reliability indicator *FVC* (provision of environmental flow for floodplain wetland vegetation) under four analysed scenarios in NRB sub-regions (pie diameter is proportional to the number of reaches in a sub-region).

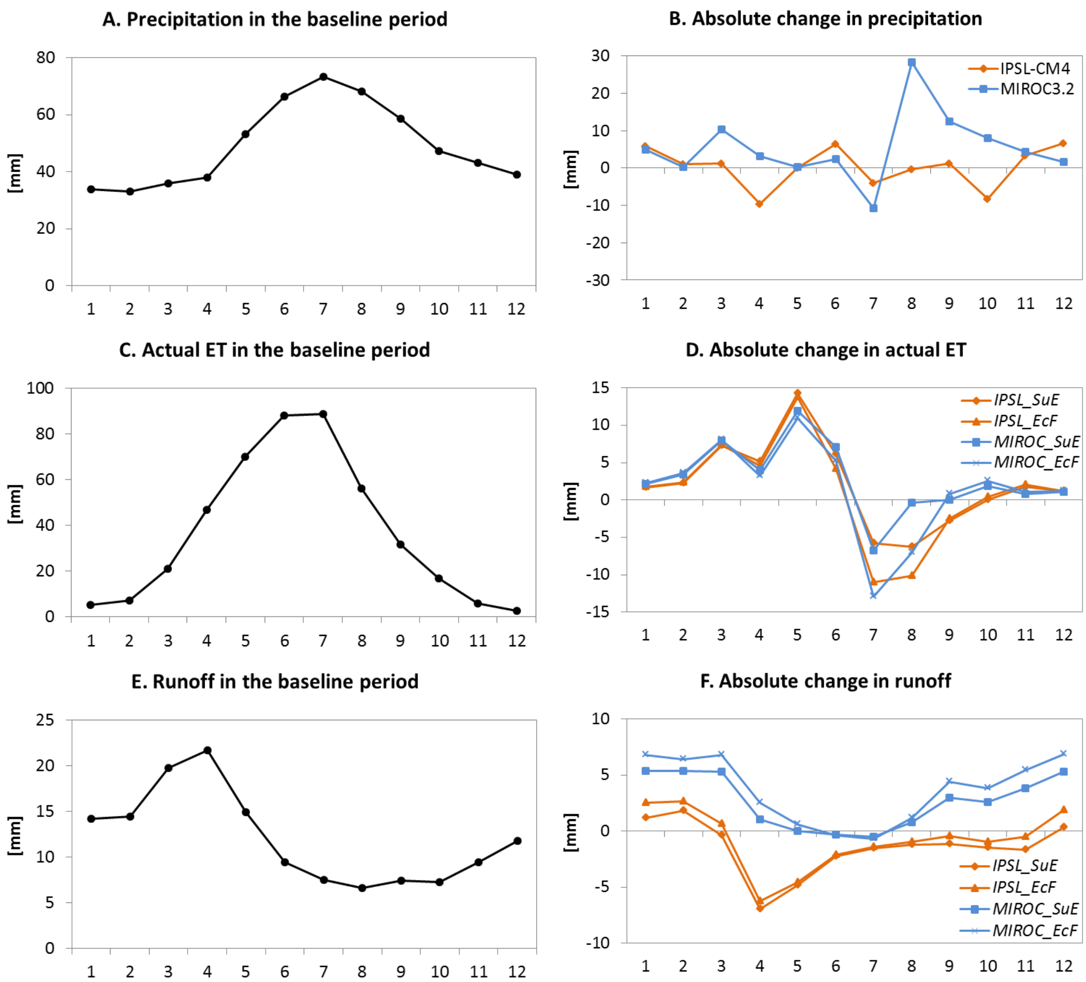

Fig. 10. Monthly distribution of basin-averaged water balance components in the baseline period and absolute changes in the future scenarios.