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Impacts of climate change and land-use scenarios on *Margaritifera margaritifera*, an environmental indicator and endangered species

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HIGHLIGHTS

- Develop a verification scheme to check stress on *M. margaritifera* based on thresholds
- Assess the impact of future climate change in conservation status of *M. margaritifera*
- Assess the impact of land-use in habitat requirements of *M. margaritifera*
- Propose conservation measures to prevent extinction of *M. margaritifera* in Portugal

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ABSTRACT

In this study, we assess the impacts of future climate and land-use in the Beça River (northern Portugal) under different scenarios and how this will translate into the conservation status of the endangered pearl mussel *Margaritifera margaritifera* (Linnaeus, 1758). This species is currently present in several stretches of the Beça River that still hold adequate ecological conditions. However, the species is threatened by projected declines in precipitation for the 21st century, with implication on the river flows and water depths that might decrease below the species requisites. This situation could be especially critical during summer conditions since the ecological flows may not be assured and several river stretches may be converted into stagnant isolated pools. The habitat connectivity will also be affected with reverberating effects on the mobility of *Salmo trutta*, the host of *M. margaritifera*, with consequences in the reproduction and recruitment of pearl mussels. In addition, human-related threats mostly associated with the presence of dams and an predicted increases in wildfires in the future. While the presence of dams may decrease even further the connectivity and river flow, with wildfires the major threat will be related to the wash out of burned areas during storms, eventually causing the disappearance of the mussels, especially the juveniles. In view of future climate and land-use change scenarios, conservation strategies are proposed, including the negotiation of ecological flows with the dam promoters, the replanting of riparian vegetation along the water course and the reintroduction of native tree species throughout the catchment.

1. Introduction

The occurrence of droughts in the early 1990s has resulted in very poor harvests and water shortages in Mediterranean regions, exposing the susceptibility of these areas to climatic extremes (Karas, 1997). The effects of the ongoing global warming can be observed in many terrestrial, freshwater and marine species that have shifted their geographic ranges, seasonal activities, migration patterns, abundances, and species interactions, namely: freshwater fishes (Regier and Meisner, 1990), plants (Araujo et al., 2004; Lemieux and Scott, 2005), mammals (Burns et al., 2003), small birds (Wilby and Perry, 2006), and macroinvertebrates (Bonada et al., 2007). According to McLaughlin et al. (2002) as well as to Pounds et al. (2006), these changes in climate may have already caused several species extinctions. The IPCC (2014)

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corroborates this idea, stating, with high confidence that under projected climate change during and beyond the 21st century many terrestrial and freshwater species faces increased extinction risk once this phenomenon interacts with other stressors, such as habitat modification, over-exploitation, pollution, and invasive species.

In general, studies on climate or land-use impacts on the conservation of aquatic biota are focused on a few vertebrate species selected on the basis of their geographical range, ecological relevance or current conservation status. Freshwater invertebrates received much less attention but inside this group freshwater mussels are among the species frequently used in those studies or programs, because they play an important role in the ecosystem, with some species being classified as indicator or umbrella species, and are one of the most endangered groups of animals on the planet (Bogan, 2008; Galbraith et al., 2010; Geist, 2010; Howard and Cuffey, 2006; Lopes-Lima et al., 2014; Skinner et al., 2003; Sousa et al., 2013). Due to its large size, sedentary, long life span and variable sensitivity to environmental contaminants, freshwater bivalves can also be very useful as indicators of ecological integrity and as sentinels of environmental perturbation (Farris and Van Hassel, 2006). This is also the case of the freshwater pearl mussel *Margaritifera margaritifera* (Linnaeus, 1758) which only occurs in pristine oligotrophic waters, being very sensible to human perturbation. In fact, the poor conservation status attained by freshwater mussels are related to human activities that include habitat loss and fragmentation (e.g. river channelization and presence of dams), pollution (e.g. nutrients, heavy metals, endocrine disruptors), overexploitation (e.g. use of shells and pearls), introduction of invasive species and climate change (Lydeard et al., 2004; Strayer et al., 2004). In addition, freshwater mussels have a specialized larva, the glochidium, that needs fish as hosts in order to complete their life cycle, which further complicates the conservation of these species.

The freshwater pearl mussel (*M. margaritifera*) is one of the most threatened freshwater bivalves worldwide (Geist, 2010). This species is a long-lived bivalve mostly occurring in cool running waters of the Holarctic region. Since the 1900s, this species declined by more than 90% in Europe and became rare or even disappeared in many European countries, including Portugal (Bauer, 1988; Buddensiek, 1995; Frank and Gerstmann, 2007; Reis, 2003). This situation triggered a conservational response and the freshwater pearl mussel is currently protected internationally by the Bern Convention (Annex III) and the European Commission Habitats Directive (Annex II and V), being listed as “Endangered” globally and as “Critically Endangered” in Europe by the IUCN Red List of Threatened Species (Cuttelod et al., 2011; http://www.iucnredlist.org).

In Portugal, the freshwater pearl mussel is present in several rivers, namely: Beça, Cávado, Mente, Neiva, Paiva, Rabaçal, Terva and Tuela (Reis, 2003; Varandas et al., 2013). In all rivers the freshwater pearl mussels were found in unspoiled stretches, located away from the major human settlements. The evidence of recent juvenile recruitment in the Rabaçal, Tuela, Paiva, Mente and Beça Rivers, make these water courses extremely important for the conservation of *M. margaritifera* in the south of Europe. However, irrespective of being well preserved in some stretches the habitat requirements for the species are currently being threatened by global climate change and various local anthropogenic pressures (Varandas et al., 2013). In the Iberian Peninsula, the species is already at the southern edge of its distribution and any changes in the temperature may become problematic (Sousa et al., in press). On the other hand, when subject to extreme climatic events, such as large return-period droughts or floods, high mortalities may occur (Hastie et al., 2003; Sousa et al., 2012). Future climatic scenarios for the Mediterranean basin over the next 50–100 years predict an increase in the mean air temperature (between 1–5 °C) and extreme events frequency accompanied by a decrease in the annual precipitation (IPCC, 2014). According to this Report (IPCC, 2014) the regional risks from climate change particularly in southern Europe include: 1) “increasing water restrictions”; 2) “significant reduction in water availability from river abstraction and from groundwater resources, combined with increased water demand (e.g., for irrigation, energy and industry, domestic use) and with reduced water drainage and runoff as a result of increased evaporative demand”; and 3) “... increasing risk of wildfires”.

In addition to climate change, Portuguese *M. margaritifera* populations (in the Beça, Terva, Rabaçal, Mente, Tuela, Neiva and Paiva Rivers) are also subject, at present, to other human threats such as the construction of dams for irrigation or hydroelectric power generation, changes in the river channel, water abstraction, disappearance or reduction of *Salmo trutta* populations that function as host for the larva, organic pollution by domestic effluents and the changes in river water quality during the period of the first rains after the occurrence of forest fires (Reis, 2003; Sousa et al., in press; Varandas et al., 2013).

A review of the literature disclosed a great number of studies on the *M. margaritifera* focused on the analysis of distribution, abundance and structure of the populations, as well as on habitat and water quality characterization (Geist, 2010). It also revealed that impact studies are much scarcer and that studies projecting future scenarios are even rarer. This study brings first elements to fill this gap by investigating the future impacts that global climate change and local anthropogenic pressures will exert in one *M. margaritifera* population located at the southern edge of the species distribution (the Beça River in Portugal). In short, the impacts are evaluated using a verification scheme based on a comparison among actual/future water quality/river flow conditions and corresponding environmental thresholds required for *M. margaritifera*. Given the scenarios of temperature increase (between 1–5 ° C) for the region under consideration in the relatively near future, it is intended with this study to predict the future environmental conditions in Beça River by using several modeling approaches. For this, we use as target species *M. margaritifera* not only because it is a flagship species, lying in danger of extinction, but also because it has a very peculiar life cycle since it requires a host species. Thus, starting from the worst scenario of temperature increase it is expected that both species (parasite and host) may disappear in the Iberian Peninsula because the ecological requirements of the species cannot become satisfied. Results will be discussed in light of the current knowledge on the biology and ecology of *M. margaritifera*. This information will be essential to design future management measures devoted to the conservation of an important endangered and indicator species at the southern edge of its distribution range.

2. Materials and methods

2.1. Study area

The Beça River, a tributary of the Tâmega River (Fig. 1), is located in northern Portugal, a humid Mediterranean region (Temperate Mediterranean with continental influences). With a total length of 55.2 km and a catchment of 345 km^2 it drains a mountainous area where the altitudes vary within 190–1270 m and the average hillside inclinations reach 11.7 ± 7.6°. Land use and occupation assessed by the 2006 Corine Land Cover inventory (Caetano et al., 2009; http://www.eea.europa.eu), available at http://www.dgterritorio.pt, showed that the region is dominated by semi-natural areas (45%), agricultural areas that include non-irrigated arable land, pastures and heterogeneous agriculture areas (32%), and forests (23%). The occupation in the headwaters and middle sector of the basin is characterized by shrubs where the relief is craggy and by dry farming areas, pastures and natural grasslands in the valleys surrounding the local villages. The downstream sector is used for wood production, being occupied by large and continuous spots of *Pinus pinaster* forests. A significant portion of these woodlands was destroyed by fire in the last decade. According to the Institute for the Conservation of Nature and Forests (http://www.icnf.pt), within the period 2000–

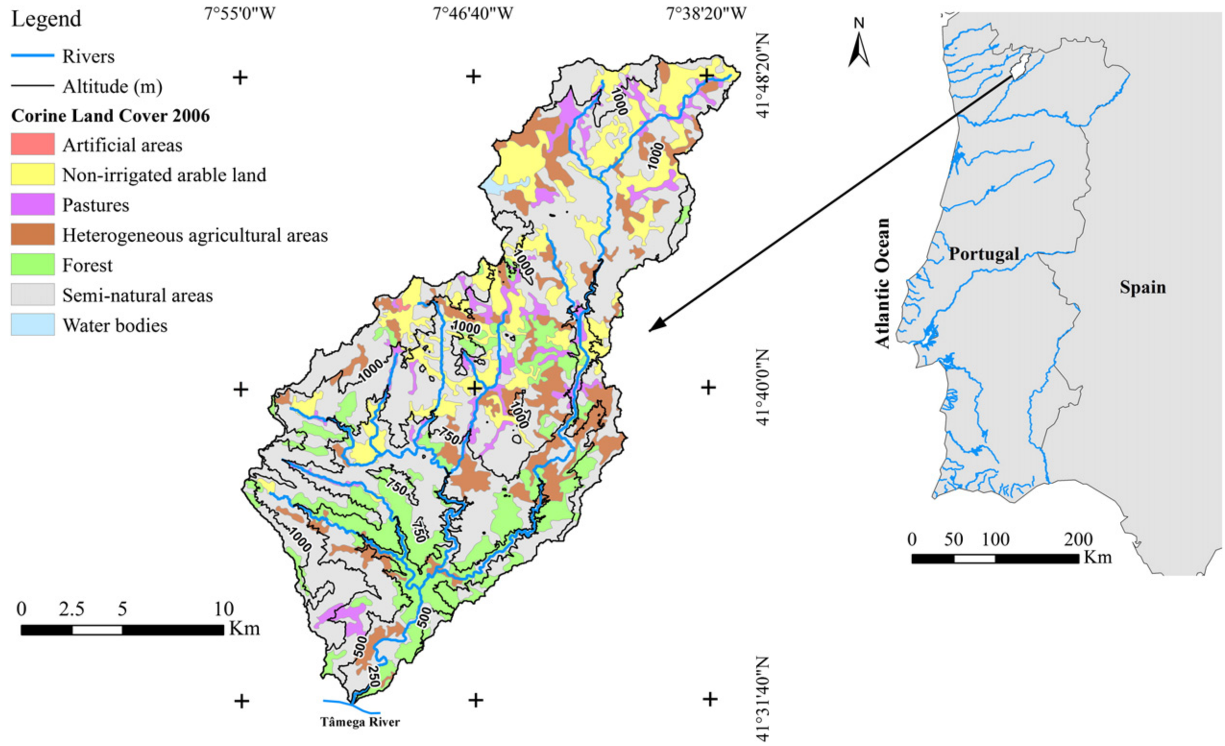

Fig. 1. Geographic location, elevation contour lines, drainage network model and dominant land uses and occupations in the Beça River basin. The land uses and occupations were compiled from the 2006 Corine Land Cover inventory.

2008 the burned areas were located mainly in the central belt and to the northeast and southwest corners of the basin (Fig. 2). The human socio-economic activities are the most probable cause of wildfires, but fires in the region increase with extreme weather conditions and land use changes with concomitant disturbances in landscape structure and function (Moreira et al., 2001; Pereira et al., 2005, 2011, 2013; Trigo et al., 2006; Viedma et al., 2006).

The Beça River basin catches an average precipitation of 1440 $mm \cdot yr^{-1}$, with a range of 650–2400 $mm \cdot yr^{-1}$. The records of average temperature and annual precipitation pertaining the period 1978–2006, but discarding the anomalous years of 2000 and 2001, indicate overall heating and drying trends of $+0.78$ $°C \cdot decade^{-1}$ and -300 $mm \cdot decade^{-1}$, respectively (Santos et al., 2014). These records are illustrated in Fig. 3, which in addition reveals a decrease in the river flows at rates proportional to the rainfall decrease.

The riparian vegetation is well-preserved in the Beça River basin, being dominated by *Fraxinus angustifolia*, *Alnus glutinosa*, *Salix atrocinera* and *Betula pubescens* (Varandas et al., 2013). After 1996, the construction of three dams (Fig. 2) for irrigation and hydropower generation resulted in a regulation of river flows and in a loss of connectivity.

2.2. Water quality data

The ecological status of Beça River was evaluated by water quality data available at the Portuguese Water Institute (http://snirh.apambiente.pt/). These data were assembled from the water quality station number 04J/11, which is located a few kilometers upstream the river mouth (Fig. 4). The water quality station is located in a lotic system and the Bragadas dam is located to the north of this station. The minimum, maximum and average values of relevant physicochemical parameters within the period 2000–2009 are depicted in Table 1. This timeframe covers the station's complete record of quality data. For the sake of evaluating the impact of wildfires (Fig. 2) in the quality of river water, a subset of values was added to the table, representing solely the summer values (June, July and August). A more recent record, assessed by Varandas et al. (2013), was also included for comparison. In this case, the purpose was to investigate the impact of the impoundment in the quality of river water, because the quality station is located downstream and the Varandas et al. (2013) measurement sites were located upstream the Bragadas dam (Fig. 4). The evaluated parameters were checked against the standards of INAG — the Portuguese Water Institute, which rank surface water quality as A (excellent), B (good), C (reasonable), D (bad) and E (very bad).

2.3. Climate change settings, models and data

Future hydrologic scenarios in the studied catchment were based on simulations of climatic variables (precipitation and temperature) using the HadRM3P and HadAM3H models, which are Regional and Global Circulation Models, respectively (Buonomo et al., 2007; Pope et al., 2000). These models were selected for this study because they are both part of the European project PRUDENCE (Christensen et al., 2002), which comprises performance analyses on these simulations (e.g., Christensen and Christensen, 2007; Christensen et al., 2007; Frei et al., 2006; Jacob et al., 2007), and because they resulted well in other studies carried out in Portugal (Gouveia et al., 2011; Ramos et al., 2011). To avoid preconception in the interpretation of results, two emission scenarios were modeled: B2 and A2, which forecast less and more severe impacts on future climate, respectively.

The climate database used in this study was composed of observed and simulated data. Observed data included maximum air temperature (*Tmax*), minimum air temperature (*Tmin*) and precipitation (*P*), all measured at the weather station of Montalegre (Fig. 4). The 1961–1990 period was elected as 30-year climatological reference because the temperature and precipitation records were essentially complete. The simulated data comprised air temperatures at 2 m (*T2m*) and precipitations at a horizontal resolution of approximately 50 km × 50 km, for the grid cell closest to the Montalegre weather station. Projections of these two climatic parameters were obtained from the PRUDENCE project website (http://prudence.dmi.dk). The simulated data also included the *adeha* simulation for the recent-past climate conditions (control period 1961–1990) and the *adhfd* and *adhfa* simulations,

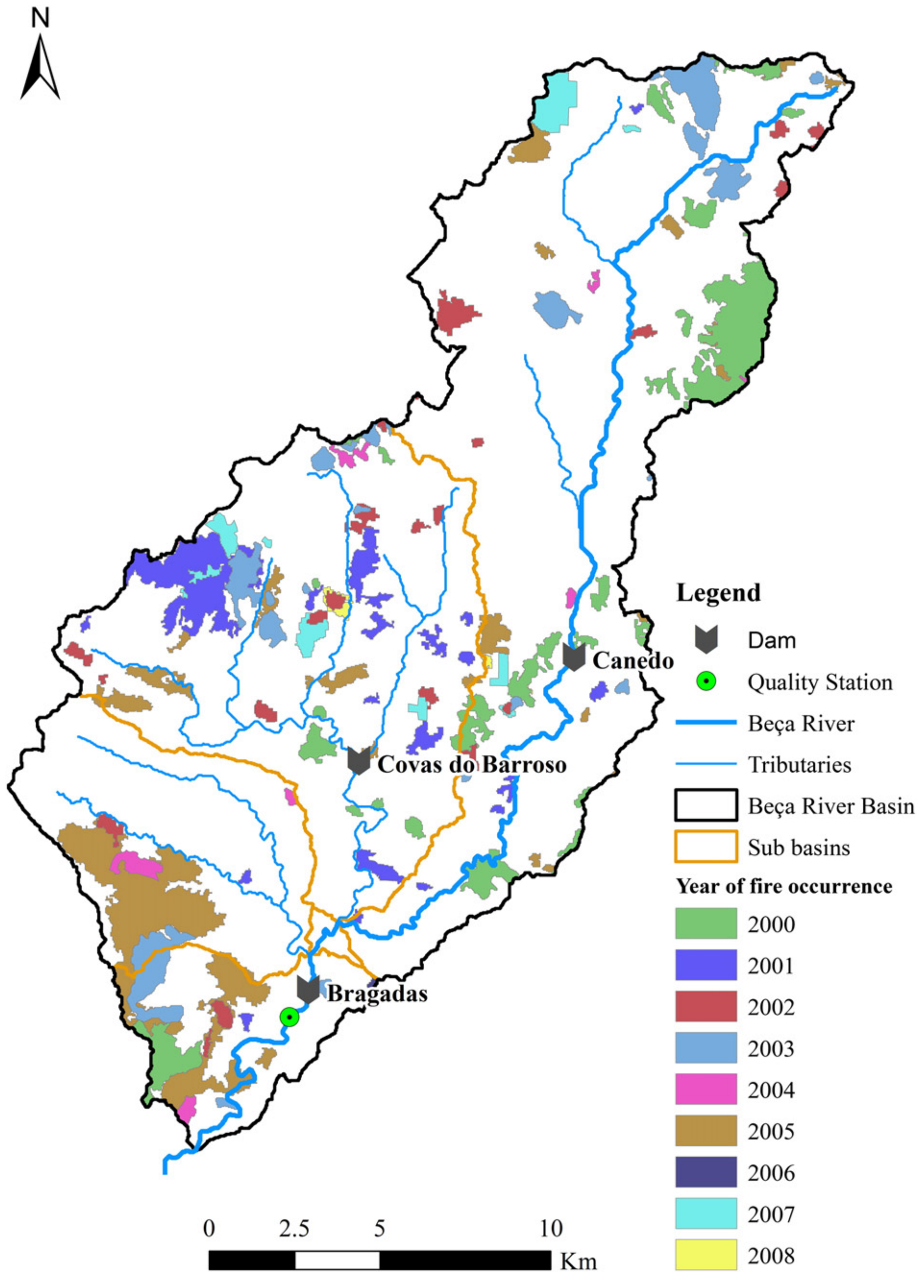

Fig. 2. Spatial distribution of burned areas, per year, between 2000 and 2008, in the Beça River basin.

respectively, for the B2 and A2 SRES scenarios (Nakicenovic and Swart, 2000) focused at the end of the 21st century (climate change period 2071–2099).

The HadRM3 simulations for the Iberian Peninsula reveal precipitation and temperature biases, which are smaller in winter and larger in summer (Jacob et al., 2007). When the aim is to forecast hydrological processes (present case), numerical simulations of climatic variables cannot be used without some form of data processing to remove the existing biases (Christensen et al., 2008; Sharma et al., 2007). So, the errors coming with the HadRM3 simulations had to be corrected in this study. Firstly, the means of temperature and precipitation in each month were calculated considering the 30-year periods of observed and simulated data. Secondly, for temperature a correcting factor was defined for each month as the difference between the means calculated for the control scenario (*adeha*) and observed data. Subsequently, these factors were subtracted from the B2 (*adhfd*) and A2 (*adhfa*) values to obtain the corrected simulations. Thirdly, for precipitation a correcting factor was defined for each month as the ratio between the means calculated for the observed data and control scenario. Subsequently, these factors were multiplied by the original B2 and A2 values to obtain the corrected simulations.

2.4. Watershed modeling

The river flows resulting from changes in temperature and precipitation in the future, as predicted by the climatic models (HadRM3P and HadAM3H), were simulated by the Mike Basin software (DHI, 2008; Madsen et al., 2002), and maps derived therefrom (e.g. Fig. 4) were produced in the GIS software ArcMap (ESRI, 2010). These integrated river-planning (Mike Basin) and broad GIS (ArcMap) computer packages have been widely used for the simulation of hydrological processes and appraisal of environmental phenomena at the catchment scale, especially in the most recent years following the massification of GIS application in scientific research (e.g. Pacheco, 2013, 2014; Pacheco and

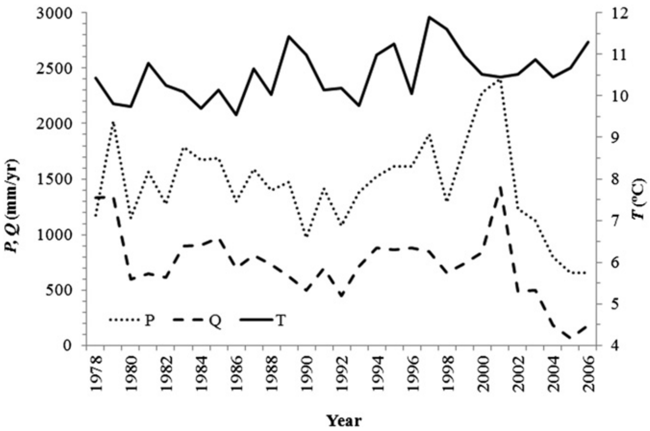

Fig. 3. Evolution of climatic parameters (precipitation — P and temperature — T) and of river flows (Q) in the Beça River basin within the 1978–2006 period. The years of 2000 and 2001 are anomalous as regards the values of P.
Adapted from Santos et al. (2014).

Landim, 2005; Pacheco and Sanches Fernandes, 2013; Pacheco and Van der Weijden, 2012a, 2012b, 2014a, 2014b; Pacheco et al., 2013, 2014, 2015; Sanches Fernandes et al., 2010, 2011, 2012, 2013; Valle Junior et al., 2014a, 2014b, 2015). In the study area, Mike Basin has previously been used with daily discharge rates to estimate river flow components within the period 1978–2006, namely the overland flow, inter and base flows (main results available in Santos et al., 2014).

In the Mike Basin software, the rainfall-runoff modeling is performed by the NAM module, which is an algorithm suited to simulate hydrologic processes at the catchment scale with moderate input data requirements (Ireson et al., 2006). In the first stage, it is necessary to assemble a watershed database, composed of graphical elements connected to alphanumeric data. In this study, the graphical elements comprised a Digital Elevation Model acquired to the Military Geographic Institute (IGEOE; http://www.igeoe.pt). The alphanumeric data were time series gathered from records available at the Portuguese Water Institute (http://snirh.apambiente.pt/) and referring to the stations plotted in Fig. 4, as well as information on the vegetation cover compiled from the Corine Land Cover inventory of 2006 (GIS database available at http://www.dgterritorio.pt). In the second stage, the watershed database had to be calibrated. The NAM model includes an automatic calibration procedure based on the comparison between measured and estimated time series of discharge rate data. The goodness-of-fit measure used in the automatic calibration of NAM results was the Nash–Sutcliffe coefficient (Nash and Sutcliffe, 1970). Calibration is considered satisfactory when this coefficient is larger than 0.6. In the study by Santos et al. (2014), carried out in the Beça River basin, the Nash–Sutcliffe index was reported to be 0.75, meaning that the NAM rainfall–runoff model was able to accurately simulate the relevant hydrologic processes in that basin. In the last stage, the NAM algorithm has to run for a specific record of climatic data to obtain estimates of the flow components. In this study the climatic data for the NAM module pertains to precipitations and temperatures predicted by the climate change scenarios B2 and A2.

2.5. Flow velocities and water column depths

Flow velocities and water column depths were measured at seven sites where the presence of freshwater pearl mussels was confirmed by Varandas et al. (2013). These river flow parameters were measured with a flowmeter every 0.5 m along river sections, at right angles of the corresponding margins. The location of the measurement sections is illustrated in Fig. 4. The data were assessed in September 2013, when the river flows were at minimum values, with the purpose of being used as reference data. Additionally, water column depths were predicted for the future using the climate change (HadRM3P and HadAM3H) and the rainfall–runoff (Mike Basin's NAM) models. Firstly, temperature and precipitation were corrected for bias and then used in the NAM algorithm to simulate daily stream flows within the period October 2071 to September 2099, under the B2 and A2 emission scenarios. The simulated flows were based on the same automatically calibrated NAM parameters and optimized values as those used by Santos et al. (2014). Secondly, the daily stream flows were converted into monthly averages, from which water column depths were estimated by:

$$H = \frac{Q}{V \times L} \tag{1}$$

where H (m) is the depth, Q ($m^3 \cdot s^{-1}$) is the simulated stream flow averaged within the month, V ($m \cdot s^{-1}$) is the average flow velocity measured at the river section during the reference period and L (m) is the corresponding section width. The calculated H values refer solely to the month of September, for which the reference velocities and section widths are usable as proxies. The column depths were estimated within the evaluated years (2071–2099) and then compared with the homologous values measured in September 2013 for the sake of investigating eventual changes.

3. Results

3.1. Water quality

Since 1998 a significant portion (>40%; Santos et al., 2014) of river water reaching the Bragadas dam is diverted to an adjacent watershed, the Tâmega River basin. As a consequence of prominent water retention, river water downstream the dam became scarce during the summer months, forming small pools dispersed along the water course. In these pools *M. margaritifera*, as well as its host (*S. trutta*), will barely keep biological conditions for survival. This conjecture is well supported by verification in the field and indirectly by the available river water quality data. In the later case, notwithstanding the good conditions of Beça River water around the quality station, the habitat requirements for *M. margaritifera* were never met between 2000 and 2009, in any of the analyzed parameters (Table 1). Conversely, upstream the Bragadas dam and beyond the influence of the reservoir, where the water quality is (mostly) excellent, the habitat requirements for *M. margaritifera* were satisfied in all parameters during the year of 2011, being its presence confirmed by Varandas et al. (2013). The same authors also observed that the benthic index of biotic integrity assembled in that study shows a Beça River in excellent ecological conditions.

An additional threat to the conservation of freshwater pearl mussels in the Beça River is forest fires. A detailed inspection of Fig. 4 reveals a total of 7563 ha of forest land burned by wildfires in the period 2000–2008. It is worth noting, however, that the area in the Beça River basin effectively affected by the forest fires was just 6613 ha (19% of the basin area) because 947 ha have burned more than once. The years of 2000 and 2005 were particularly dramatic, because wildfires covered 4.7% and 7% of the total catchment area, respectively. A good relationship between wildfires and river water quality was established since major peaks in electric conductivity (62.4 $\mu S \cdot cm^{-1}$ in 2005; Fig. 5a) and total phosphorus (0.31 mg $P \cdot L^{-1}$ in the same year; Fig. 5b) are coupled with major peaks representing the percentage of burned area.

3.2. Climate change

In order to inspect the ability of HadRM3 to reproduce past climatic conditions as well as to project climate changes into the future, a comparison was made between 30-year averages of air temperature and precipitation, involving: (i) observed and simulated values in the

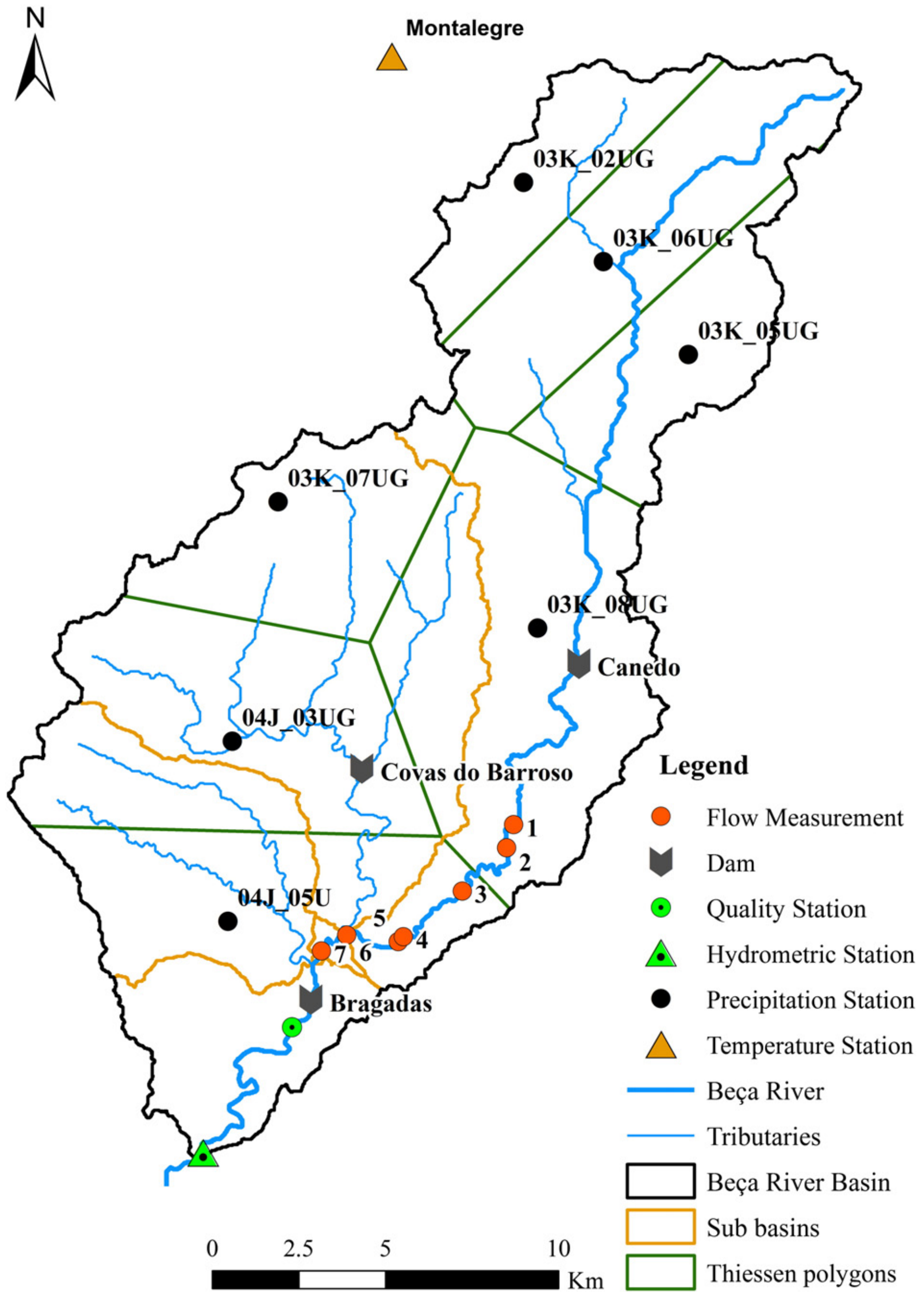

Fig. 4. The Beça River basin with the main water course, tributaries and sub-basins. Location of stations used for the monitoring of temperature, precipitation, river flow and water quality. Thiessen polygons used in the calculation of weighted average precipitation from discrete values measured at the udometric stations.
Adapted from Santos et al. (2014).

1961–1990 period (control scenario); (*ii*) simulated values in the 1961–1990 and 2071–2099 periods, considering in the latter case the IPCC emission scenarios B2 and A2.

The consequence of using simulated data without bias correction is that air temperatures for the control and climate change scenarios are larger than for the observed data, in all months (Fig. 6a). The observed and control curves are relatively close to each other and depart from the B2 and A2 curves, which are also close to one another. The departures between curves are not constant throughout the year but show a high seasonal variability. The largest shifts occur from June to September and the smallest ones from December to March. For example, the differences between observed and simulated A2 temperatures reach a maximum of 10.1 °C in August and a minimum of 3.2 °C in March. Comparisons between the B2 and A2 scenarios disclosed much smaller differences (<3 °C), the largest ones referring to the June–September period. The procedure used to correct bias forced the control scenario predictions to reproduce exactly the observed values and caused a generalized decrease of air temperatures in the climate change scenarios (Fig. 6b). After correction, the differences between observed and simulated data ranged between 1.9 °C in March and 4.7 °C in July, when scenario B2 is considered, and between 2.1 °C in March and 6.5 °C in September when the investigated scenario is A2.

Before correction for bias, the control scenario precipitations were lower than the measured values, with the exception of July and August. On the other hand, the B2 and A2 precipitations were smaller than the control scenario values, with exceptions in the late autumn to early spring months (Fig. 6c). Similarly to temperature, differences were not uniform throughout the year. The smallest differences in relation to observed data are expected to occur in July and August, with values ranging between 2.6 and 12.4 mm, while the largest differences are

Table 1
Physic-chemical parameters of the Beça River water measured at the 04 J/11 water quality station between March 2000 and June 2009. Data measured by Varandas et al. (2013) between June and August 2011. In the first case, the measurement site is located downstream the Bragadas dam, in the second case upstream that structure (Fig. 4). Classification of surface water quality based on standards defined by the Portuguese Water Institute. This classification comprises five quality classes: A — excellent; B — good; C — reasonable; D — bad; E — very bed. Habitat requirements for the sustainability of *M. margaritifera*, defined by Varandas et al. (2013): (*i*) minimum or maximum thresholds, depending on the parameter; (*ii*) status of Beça River water as regards the fulfillment of those requirements, downstream (D) and upstream (U) the Bragadas dam. The status is either Yes — respects the habitat requirement or No — does not respect the habitat requirement.

Parameter	Downstream the Bragadas dam						Upstream the Bragadas dam			INAG classification 2000–2009/2011	Habitat requirements for *M. margaritifera*	
	2000–2009			June, July and August 2000–2009			June, July and August 2011				Threshold	Status (D/U)
	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max			
Temperature (°C)	2	12.9	24.1	14	19.4	24.1	11.2	13.2	14.8	Not applicable	<23	No/Yes
Conductivity (μS cm^{-1})	14	25.5	62.4	19.4	27.8	47	27.1	29.4	31.6	A	<40	No/Yes
pH	5.6	6.5	7.4	6.1	6.7	7.2	6.4	6.6	6.8	B	≤7	No/Yes
O_2 (mg O_2 L^{-1})	8	10.1	14.6	8	8.9	10.8	9	9.5	10.1	Not applicable	>9	No/Yes
O_2 (%)[a]	68	94.8	123.5	90.4	97.1	106.3	92.2	96.7	101.2	C/A	>90	No/Yes
Total phosphorus (mg P L^{-1})	0.01	0.05	0.31	0.01	0.05	0.27	–	–	–	C/–	<0.10	No/Yes
Orthophosphate (mg P L^{-1})	–	–	–	–	–	–	0.02	0.04	0.04	A	<0.10	–/Yes
BOD_5 (mg O_2 L^{-1})	0	1.8	4.2	0.5	1.9	3	1.2	1.4	1.5	B/A	<1.50	No/Yes
Ammonium (mg NH_4^+ L^{-1})	0.01	0.05	0.42	0.01	0.06	0.42	0.04	0.05	0.05	A	<0.10	No/Yes
TSS (mg L^{-1})	0	3.7	23.2	0	3.5	7.2	17.9	19	20	A	≤20	No/Yes
Nitrate (mg NO_3^- L^{-1})	–	–	–	–	–	–	<2	<2	<2	A	<2	–/Yes

(–) no data available.
[a] The data were available only between September of 2001 and December of 2005.

forecasted for April (72.3 mm) and October (64.6 mm), depending on the climate change scenario. The curves representing the B2 and A2 scenarios are very similar, showing the largest shift in January (31 mm). After bias correction, control scenario precipitations were forced to coincide with the observed data. This forcing caused an increase of precipitations simulated for the B2 and A2 scenarios and consequently a closer similarity between present-day and future seasonal variability (Fig. 6d). Nevertheless, with a few exceptions, simulated precipitation is higher than the observed values from late autumn to early spring (November to March) and lower throughout the rest of the year. The maximum differences are expected in April (+54.8 and +69.7 mm for B2 and A2, respectively) and the minimum differences in November (−4.3 mm, B2) or March (−0.5 mm, A2). When the differences between projected (scenarios B2 and A2) and actual (control scenario) precipitations are assembled by season (Table 2), one sees that (*a*) they are positive in winter and negative in the other seasons which means that future winter seasons are expected to be rainier than present-day winters; (*b*) A2 differences are larger than B2 counterparts. On the annual basis, the differences are B2 (81.3 mm) > A2 (61.7 mm).

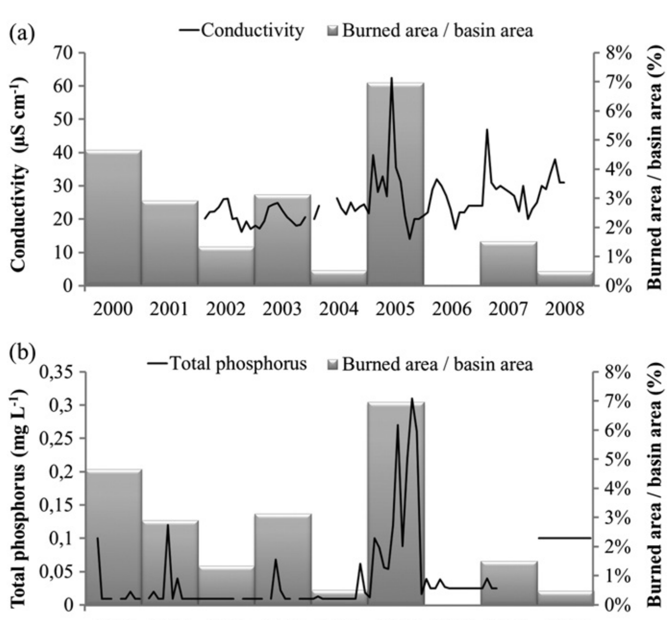

Fig. 5. Records of electric conductivity and of total phosphorus dissolved in surface water (solid lines in the *a* and *b* plots), and of areas burned within the Beça River basin (shaded columns in the same plots), between 2000 and 2008.

3.3. Runoff

Using Mike Basin NAM, Santos et al. (2014) estimated Beça River flows within the period 1978–2006. In forthcoming sections, this timeframe is referred to as control period. Considering the same calibrating parameters and optimized values as adopted in that study, the NAM model was used to forecast river flows within the 2071–2099 period, called the climate change period. The results are illustrated in Figs. 7 and 8, which represent the monthly and yearly bases, respectively.

On the monthly basis (Fig. 7), the differences between climate change and control runoff values generally follow the tendency of future increases in precipitation during the wet seasons and decreases during the dry seasons (see Figs. 6d and 7). When considering the emission scenario B2, the runoff increase is likely to occur in December and February, by 12% and 2%, respectively. Under the A2 scenario the runoff increase is projected for December to March, with values ranging between 2% in March and 18% in February. The major runoff decreases are likely to occur between April and June and in October, in both scenarios. In scenario B2 these large decreases may vary between −30% in June and −37% in October, while in scenario A2 they are expected to be somewhat larger varying between −30% in June and −44% in October. The months of July and August are thought to register much smaller runoff decreases: −20% and −18% in scenario B2; −18% and −16% in scenario A2.

As regards the annual variation (Fig. 8), the projections reveal a pronounced decrease in runoff between 2071 and 2099, with the exception of a few years, namely 2072, 2075, 2079, 2083 and 2094. The runoff decrease may become more marked over the years but fluctuations are expected to be large in both climate change scenarios. The larger decreases are likely to occur in scenario A2 and may reach more than 50% by the end of the climate change period.

3.4. Water depths

Average water depths in sections 1 to 7 (Fig. 4) were simulated for the control and climate change (B2 and A2) scenarios, and the

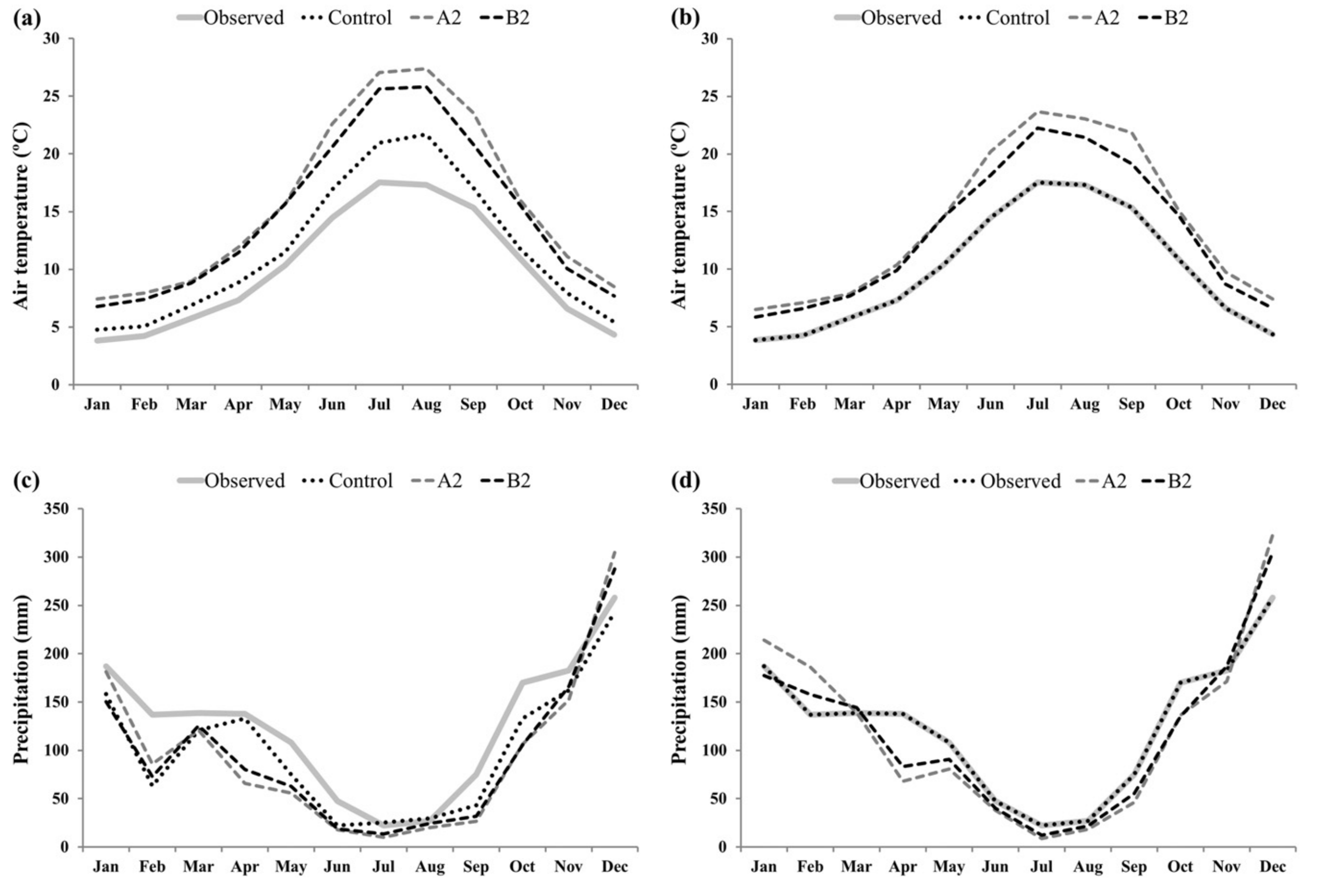

Fig. 6. Monthly distributions of air temperature (plots *a*, *b*) and precipitation (plots *c*, *d*) in the Montalegre weather station. The various lines represent distributions of measured data, and of simulations covering the control (1961–1990) as well as the climate change (2071–2099) periods, the latter being projected under the B2 and A2 emission scenarios. In the plots (*a*) and (*c*), simulations do not account for bias correction of the climatic parameters. In the plots (*b*) and (*d*) they do.

minimum values obtained for the month of September were plotted in Fig. 9. For the sake of simplicity and comparison, the average water depth measured along the same river sections in September of 2013 was also represented in that figure. In the Beça River, the minimum water depth for the conservation of *M. margaritifera* was reported to be between 0.2–0.4 m (Varandas et al., 2013) and, in the Rabaçal and Tuela Rivers, Teixeira et al. (2010) found that the optimal depth was between 0.3 and 0.4 m. Given the proximity of the value ranges found in both studies, it was decided to consider as optimal depth to the species, the average class (0.3 m) observed by Varandas et al. (2013) which corresponds to the minimum class value observed by Teixeira et al. (2010). During the control period, this threshold was always exceeded in sections 2, 3, 5 and 6, but was not attained in sections 1 (once, in the year of 2005), 4 (twice, in the years of 2004 and 2005) and 7 (in 2005). The measurements of September 2013 reveal a slight aggravation of this situation because section 2 joined the group of sites where the average water depth is lower than 0.3 m at least once during the control period. The forecast is even worse when climate change scenarios B2 and A2 are considered, because only sections 3 and 5 will remain with water depths sufficient for the survival of *M. margaritifera*. On the other hand, the number of years showing depths below 0.3 m in September are likely to increase (e.g. from 2 to 3 years in section 4) and the minimum depths in that month will tend to decrease by 39.5% to 44.7%, depending on the scenario (e.g. from 0.18 m to 0.097–0.106 m in section 7). One should recall that water columns illustrated in Fig. 9 do not represent the full range of depths in the sections because only average values are indicated. In September 2013, when the in situ water depths were measured every 0.5 m along the cross sections, the maximum values varied between 0.32 m (section 7) and 0.61 m (section

Table 2
Differences between bias corrected precipitations projected for the climate change scenarios (B2 and A2) and observed in the control scenario, assembled by season and on an annual basis. The capital letters inside brackets represent the first letter of a given month (e.g. D — December).

Period	Difference between precipitation	
	B2-control	A2-control
Winter (DJF)	58.3	141.6
Spring (MAM)	−66.4	−96.5
Summer (JJA)	−22.8	−31.8
Autumn (SON)	−50.4	−74.0
Annual	−81.3	−60.7

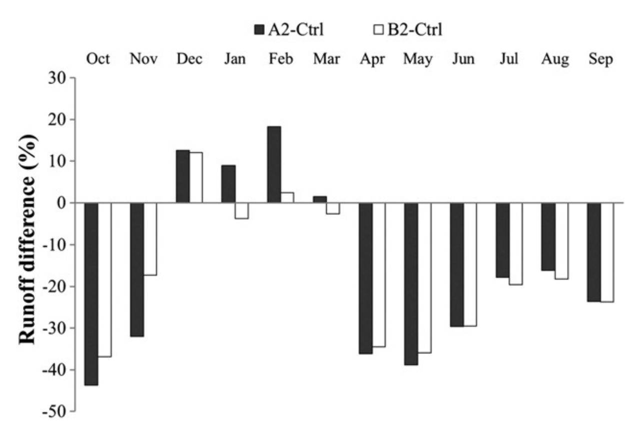

Fig. 7. Monthly distributions of runoff differences (in percent): between A2 and control simulations (filled columns), and between B2 and control simulations (unfilled columns).

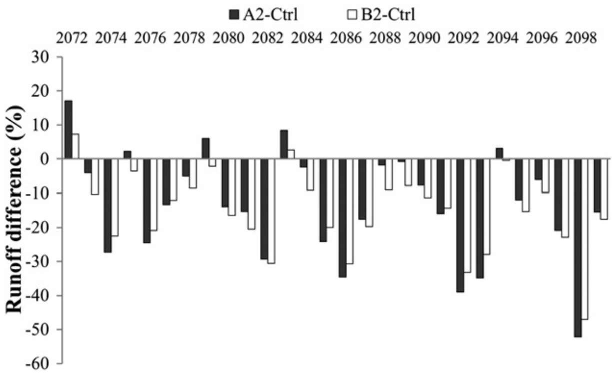

Fig. 8. Annual distributions of runoff differences (in percent): between A2 and control simulations (filled columns), and between B2 and control simulations (unfilled columns).

4), meaning that somewhere in the section the habitat requirement is still guaranteed. However, given the prediction of a drop in the water depths to half the current values in the period 2071–2099, even the maximum depths will tend to approach the critical threshold.

4. Discussion

4.1. Future climate change and land-use scenarios impacts on Margaritifera margaritifera

Climate change is one of the most important causes of biodiversity loss in freshwater ecosystems (Cosgrove et al., 2012; Sala et al., 2000), and it is expected to cause extinctions in many species in the future (Thomas et al., 2004). Species are affected by climate change because these changes modify the environmental gradients that interact with them (Galbraith et al., 2010; Hastie et al., 2003; IPCC, 2014) with consequent shifts in phenology, range and physiology (reviewed in Bellard et al., 2012).

The Mediterranean region has experienced large climate shifts in the past (Luterbacher et al., 2006) and has been identified as a hotspot in future climate change projections (Giorgi, 2006). The Iberian Peninsula is among the areas where projections of climate change impacts are more significant, particularly during summer when the ensemble average change (2071–2100 minus 1961–1990, A1B scenario) can be as higher as 5 °C in surface air temperature and −40% in annual precipitation (Giorgi and Lionello, 2008). Other projections for the end of the 21st century and under the IS92a scenario suggest a decrease of 10–50% in the ensemble mean change of annual runoff and a change in the recurrence of droughts from 100-year to just 10-year return periods (Kundzewicz et al., 2008; Lehner et al., 2005). Our results corroborate this latter scenario, since they consider that the Beça River basin will suffer decreases in runoff that can reach over 50%, given the A2 scenario.

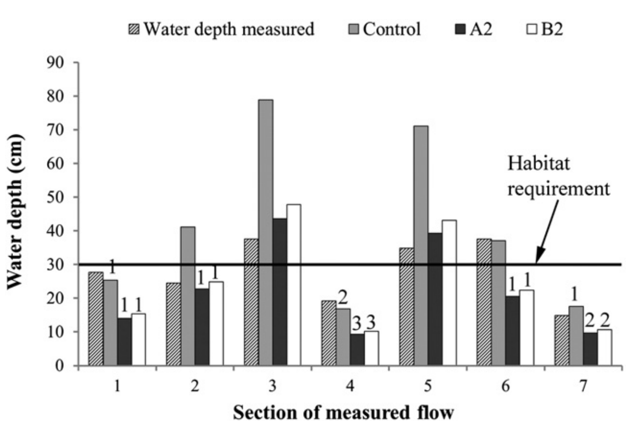

Fig. 9. Water depths measured or projected for the 7 river sections plotted in Fig. 4. The distributions refer to the month of September, the most critical to *M. margaritifera* because in this month depths are usually at minimum values and hence closer to the habitat requirement of 30 cm (Varandas et al., 2013). The simulated depths refer to the control period (1961–1990) as well as to the climate change period (2071–2099). As regards the climate change period, the plot illustrates simulations based on the B2 and A2 emission scenarios. The numbers placed above the columns indicate the number of years within the corresponding period, when the water depths were below the habitat requirement.

The ideal habitat for *M. margaritifera* is a lotic system where no discontinuities in water masses are observed. The discontinuity of water masses (habitat fragmentation) influences *M. margaritifera* because this species has a very low mobility and a clear preference for the colonization of sites near the banks. On the other hand, it affects the mobility of *S. trutta* populations, with direct consequences in the reproductive capacity of the pearl mussels (Sousa et al., in press). In the studied sector of Beça River (Fig. 4), water levels required by the species will persist in sections 3 and 5, either at present-day or future climatic conditions, because the water depths are always greater than 0.3 m (Fig. 9). However, the average depth in sections 4 and 7 (especially in September) foreshadows a possible habitat fragmentation by loss of connectivity during the summer season, since it was lower than 10 cm (Fig. 9). Such a situation can be of great concern because it will lead to the decrease in the size of the bivalve populations which can be spread to all other sections located upstream, even where the ecological requirement of depth > 0.3 m is permanently satisfied. This is probably the reason why the observed *M. margaritifera* abundances decrease from section 7 to section 2 (Table 3). The overall perspectives on habitat fragmentation are problematic because sections 4, 6 and 7 correspond to the river reaches B5 and B6 of Varandas et al. (2013) where the number of freshwater pearl mussels was the largest: 76 and 99, respectively (Table 3). In addition, pearl mussels in the Iberian Peninsula reproduce during September (Sousa et al., in press), which means that very low depths during this month may impair reproduction because will affect the presence of hosts in several areas. Apart from the physical habitat fragmentation caused by the lowering of water depths, the ongoing climate change will also lead to a reduction in water quality (IPCC, 2014). It also reports that most species (especially plants, small mammals and freshwater mollusks) will not be able to shift their geographical ranges sufficiently fast to keep up at the rates projected. Therefore, *M. margaritifera* in the Beça River is highly threatened and highly prone to extinction. Indeed, Portuguese populations are located at the southern edge of *M. margaritifera* distribution where the climate is warmer and these populations are likely to be the first to disappear in the course of future temperature increases (Sousa et al., 2013).

As regards precipitation, both climate change scenarios (B2 and A2) predicted increments for the winter months and declines for the rest of the year, forewarning an increment in the frequency and severity of extreme climatic events. According to Sousa et al. (in press), such events may be problematic for freshwater pearl mussels. Data regarding the

Table 3
Freshwater pearl mussel distribution and density in river reaches (B1–B6) studied by Varandas et al. (2013). Correspondence between the B1–B6 reaches and the 1–7 sections used in this study for the measurement of river flows and water depths (Fig. 4).

River section	River reach	Number of mussels	Mussel density (mussels·m^{-2})*
–	B1	0	0.0
1	–	–	–
2	B2	1	0.0001
3	B3	1	0.0001
–	B4	6	0.003
4 e 5	B5	76	0.010
6 e 7	B6	99	0.010

large floods of early 2010 in several Portuguese rivers described great mortalities of *M. margaritifera* (Sousa et al., 2012). Similarly, in Scottish rivers (e.g. Avon, Spey, Kerry), large floods adversely affected mussels and when exceptionally large floods occurred great mortalities occurred as well (Hastie et al., 2001, 2003). Although these stochastic events were historically rare, there is evidence to suggest that they may be occurring more frequently as a result of climate change (Black, 1996).

The drastic decline or disappearance of *M. margaritifera* populations in the north of Portugal and in Europe until the first half of the 20th century was a direct consequence of river water quality deterioration (Geist, 2010; Reis, 2003, 2006; Sousa et al., 2013). In Portugal, the species is still present in the upper reaches of river basins where human activities are reduced and water quantity and quality are good or excellent (Reis, 2003, 2006; Sousa et al., in press; Varandas et al., 2013). Among the activities threatening the freshwater pearl mussels, construction of dams may be considered prominent (Baskaya et al., 2011; Cosgrove et al., 2000; Metzeling et al., 2004). Changes include alterations in channel morphology, increased water depth, change from lotic to lentic conditions, accumulation of fine sediments, and altered seasonality of flow, temperature and oxygen (Addy et al., 2012; Vaughn and Taylor, 1999). The comparison of data collected by Reis (2003) and Sousa et al. (in press) in several Portuguese rivers shows that the construction of dams coincided with severe declines of *M. margaritifera*. In the case of this study, the presence of dams may increase the negative effects of future climate change scenarios. For example, water quality in sections of the Beça River located upstream and downstream the Bragadas dam where the *M. margaritifera* is present and absent, respectively, showed that the habitat requirements for the species amply satisfied in the sections located upstream the dam and beyond the influence of the reservoir but not in the section located downstream. Eventually, the degradation of river water quality occurred when the ecological flow was not released during the summer season and the river flows downstream the dam were converted into stagnant water filling isolated pools. In these pools, enhanced evaporation increased the conductivity, the TSS and the concentrations of ammonium, total phosphorus and BOD_5 leading to raised macrophyte growth and in turn to a decrease in the oxygen levels. Overall, the construction of Bragadas dam precluded the presence of freshwater pearl mussels downstream the weir until the confluence between the Beça and Tâmega rivers (9 km). Given the projections of climate change during drought periods, the river Beça can be converted into isolated pools that, according to Cosgrove et al. (2012) lead to increased water temperatures sometimes exceeding the critical upper limit for the survival of *M. margaritifera*. This can be especially acute for juveniles because they are more demanding in terms of habitat requirements: cool, well-oxygenated soft water, free of pollution or turbidity. At this point, it is also worth noting that *S. trutta*, the host of *M. margaritifera* in the larval stage (for a review see Geist, 2010), is also extremely sensitive to temperature increases beyond 25–30 °C (critical upper limit), as reported in Hastie et al. (2003).

Sousa et al. (in press) reported the disappearance of *M. margaritifera* from the first 4 km of a river stretch situated upstream a dam, associating the observation to lentic conditions created by the weir. However, other alterations may also occur downstream of dams, like releases from impoundments which often result in either abnormally high or low flows (Baskaya et al., 2011; Vaughn and Taylor, 1999). Layzer et al. (1993) and Hastie et al. (2003) concluded that these rapid alterations in high and low flows produce short-term near-flood or near-drought conditions which are incompatible with the maintenance of mussel populations. High water velocities can also alter the sediment dynamics, affecting both adults and juveniles (Sousa et al., in press). Sediment stability is usually considered a key factor determining the presence of pearl mussels (Geist and Auerswald, 2007). On the other hand, long periods of low flow below impoundments can result in mussel mortality due to stranding (Sousa et al., in press).

In compliment to the effects of dams on water quality, *M. margaritifera* populations are frequently affected by changes in land use and occupation (Geist, 2010; Young et al., 2001), namely by wildfires, being this situation especially important in southern European countries. An important consequence of wildfires is the increase of soil erosion with the accompanying transport of suspended fine sediments and dissolved nutrients downhill (Smith et al., 2011, 2012). The siltation and compaction of the substrate with reduction of the interstitial water oxygen levels are one of the most important reasons for *M. margaritifera* decline. Fine silts also clog the bivalve gills decreasing respiration rates (Geist and Auerswald, 2007). The large inputs of nutrients increase electric conductivity and reduce dissolved oxygen, which may affect the survival of *M. margaritifera*.

4.2. Implications for conservation

This work highlights future climatic and land-use impacts on the conservational status of *M. margaritifera* in the Beça River. Because the forecasted impacts are negative, it becomes urgent the application of conservation measures to prevent the extinction of this species in Beça River. Among the possible conservation measures, the most essential are those counteracting the existing human pressures. For that reason, one considered a number of conservation measures related to the preservation of water quality and habitat connectivity, namely: (*a*) the restoration of stream flow connectivity through the removal of obsolete weirs and the maintenance of adequate ecological flows. In this context, a negotiation with the promoter Bragadas dam to release ecological flow throughout the year is essential; (*b*) the replanting of riparian vegetation where it is absent, using native species like *Alnus glutinosa*, *Fraxinus* spp. and *Betula* spp., to create buffer strips between the river and adjacent agriculture and forest lands. This strategy, in agreement with Cosgrove et al. (2012), is one of the most important measures to prevent or reduce the severity of climate change effects on pearl mussels. Also, researches have showed positive correlations between pearl mussel beds and tree cover (Baer, 1981; Hastie et al., 2003; Varandas et al., 2013). It is believed that pearl mussels (and their host fish) benefit through the shading of watercourses once these buffers are important water quality regulators: 1) by reducing the extremes of hydrothermal fluctuations and preventing excessive algal growth, which can smother the river bed (Cosgrove et al., 2012); and 2) working out as filters of suspended sediment and dissolved nutrients transported from uphill, especially in the course of pervasive land use changes such as wildfires. The restoration could also help to reduce erosion and stabilize edge habitats through the roots of native broadleaved trees (Parrott and MacKenzie, 2000); (*c*) the replacement of pine by native species such as *Quercus* spp. and *Castanea sativa*, especially at the downstream sector of the basin where pine areas are dominant, to reduce the risk of wildfire occurrence. Apart from these specific conservation measures, institutional responses are essential to assure the long-term survival of *M. margaritifera* in the Beça River.

5. Conclusions

The investigated freshwater pearl mussel (*M. margaritifera*) is considered an indicator species with great conservational importance, mainly in Europe. Therefore, predictions about future impacts of climate change and land-use are extremely important. In the case of this study we clearly showed that reductions in water column depths below the required value of 30 cm are expected, which may impair the future survival of the species in this river. In addition, the presence of a dam clearly reduces downstream flows below the ecological thresholds. In summer, the reduced flows are not able to create a continuous film of water along the river channel, a circumstance leading to habitat fragmentation. This situation may also affect *S. trutta*, the natural host of the larval stages of *M. margaritifera*, with direct consequences in the reproductive capacity of the mussels. Finally, wildfires trigger hydric erosion downhill, which may result in siltation and compaction of the substrate. This situation may reduce the interstitial water oxygen levels, may clog the bivalve gills decreasing respiration rates, increase nitrate and phosphorous concentrations, which ultimately may affect *M. margaritifera*. The overall impact assessment forecasts the necessity and urgency of implementation of conservation measures. Indeed, considering the conservation status of this species, its complex life history, the specific habitat requirements, the imminent impact of climate change and the geographic location near the southern limit of its distribution, we strongly recommend the consideration of the Beça River as Special Protection Area (SPA) under the Habitats Directive.

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