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ORIGINAL ARTICLE

Assessing biodiversity vulnerability to climate change: testing different methodologies for Portuguese herpetofauna

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Abstract Assessing biodiversity vulnerability to future climate change is essential for developing robust adaptation strategies. A number of vulnerability assessment methodologies have been developed, from bioclimatic envelop models to more complex approaches that also consider biological traits and population status. However, the lack of comparative studies leaves the user to choose among the different methodologies without much guidance. This study applied three vulnerability assessment approaches to the Portuguese herpetofauna: (I) impact assessment approach based on bioclimatic models; (II) integrated vulnerability assessment approach, adding the evaluation of adaptive capacity to approach I; and (III) integrated vulnerability assessment and validation based on expert consultation. Results showed disagreement between the different approaches for 19 % of the species studied. Most differences were found between approach III and the two other approaches. All approaches showed advantages and limitations, the choice of a methodology being ultimately dependent on the study goals. Approach I has proven efficient to capture general vulnerability patterns. Approach II, although presenting results similar to approach I, allows for the identification of key factors affecting the species adaptive capacity and may be useful in tailoring adaptation measures. Approach III further allows us to identify knowledge gaps and to evaluate vulnerability when data availability or quality is reduced. Further, because this approach is based on an expert workshop, it has proven a perfect means to build on the vulnerability assessment results to identify indicator species and prioritize specific adaptation options.

Keywords Adaptive capacity · Biodiversity vulnerability assessment · Climate change · Bioclimatic envelope models · Climate change vulnerability indexes

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Introduction

Climate change is expected to have profound ecological impacts, likely becoming one of the greatest threats to biodiversity during this century (Millenium Ecosystem Assessment 2005; Bellard et al. 2012; Settele et al. 2014). A number of key conventions and directives determined the development of biodiversity policies across Europe (e.g. the Ramsar Convention, the Bonn Convention, the Bern Convention, the Habitats Directive and the Birds Directive). These policies are heavily oriented towards the concepts of creating nature reserves and in situ conservation, both suiting a static environment, but not conceived to cope with expected impacts of climate change where large-scale climate niche displacement is very probable (Cruz et al. 2009; Heller and Zavaleta 2009; Trouwborst 2012; García et al. 2013; Settele et al. 2014).

Although the biodiversity sector is considered prioritary in the European Union's ten-year growth strategy (Europe 2020) and by all countries developing national climate change adaptation strategies (Swart et al. 2009), the integration of biodiversity conservation measures into adaptation strategies is often still lacking (Brooker and Young 2005; Settele et al. 2014). The uncertainties involved in assessing biodiversity vulnerability are still a strong constraint to develop robust adaptation strategies and prioritize action (Pressey et al. 2007; Cruz et al. 2009; Heller and Zavaleta 2009; Bagne et al. 2011; European Union 2013; Girvetz et al. 2014).

The definition of vulnerability in the fourth assessment report of the International Panel of Climate Change's (IPCC) incorporates the concepts of potential impacts, function of exposure and sensitivity, and adaptive capacity (Fig. 1; IPCC 2007). Exposure is the nature and degree to which a system is exposed to climatic variations (i.e. changes in climate conditions such as temperature and precipitation). Sensitivity is the degree to which a system is affected by climate change. For species, this will be the product of the breath of climate conditions in which a species is known to survive. Adaptive capacity is the ability of a system to adjust to climate change. For species,

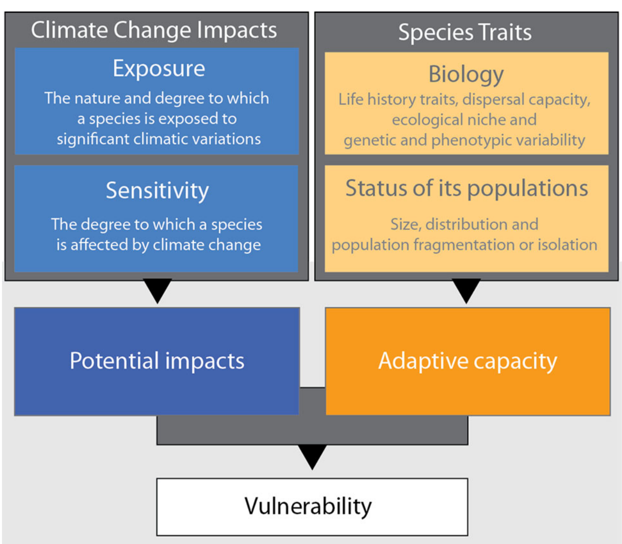

Fig. 1 Schematic representation of the factors affecting species vulnerability to climate change. Vulnerability is represented as the product of potential impacts, function of exposure and sensitivity (*blue pillar*) and adaptive capacity, function of the species biology and status of its populations (*orange pillar*). Adapted from IPCC (2007)

adaptive capacity will depend on both its specific biology (e.g. genetic or phenotypic variability, dispersal capacity) and the status of its populations, which in turn is a consequence of human pressures such as overexploitation or habitat fragmentation (European Union 2013).

The overwhelming complexity of the natural systems presents fundamental limits to modelling species vulnerability to future climate change (Pearson and Dawson 2003; Willis and Bhagwat 2009; Bagne et al. 2011). Evidence shows that climate change is already affecting species distributions and that observed changes are not just dependent on the species climate niches but also on other traits such as dispersal capacity and biotic interactions (Erschbamer et al. 2009; Devictor et al. 2012; Caldas 2014). These studies thus support the idea that any assessment to forecast species future vulnerability to climate change should include both an impact assessment and an adaptive capacity assessment. However, most assessments conducted so far have focused on biodiversity impact assessments only, mainly based on bioclimatic models (Pearson and Dawson 2003; Berry et al. 2007; Bertzky et al. 2011).

Bioclimatic models relate observed occurrences of species with historical climatic conditions and predict future potential distributions using projected changes in climate variables (e.g. maximum temperature, average annual rainfall). Thus, they assess the direct impacts of climate change (Fig. 1—blue pillar) and do not consider a number of other factors that affect species vulnerability to climate change, including species' life-history traits (e.g. survival rates and generation times), habitat suitability (e.g. habitat fragmentation), and potential indirect effects of climate change (such as interactions with predators or effects of climate change on habitats the species depend on; Fig. 1—orange pillar) (Berry 2008; Henle et al. 2008; García et al. 2013; Caldas 2014). These limitations of bioclimatic modelling can lead to incorrect estimations of species vulnerability and extinction risk (de Chazal and Rounsevell 2009; Early and Sax 2011; Urban et al. 2012).

Biological traits such as genetic variability, generation time, dispersal ability and population status affect each species' adaptive capacity and thus its vulnerability to climate change (Williams et al. 2008; Devictor et al. 2012). However, the effects of those traits on vulnerability are hard to quantify or predict and have therefore received less attention than the direct effects of abiotic factors (Williams et al. 2008; Staudinger et al. 2012). Recently, several indexes for integrated vulnerability assessments that include both potential abiotic impacts and species adaptive capacity (i.e. both the blue and orange pillars in Fig. 1) have been developed and are starting to be tested (Lankford et al. 2014; Young et al. 2014). These include vulnerability indexes based on expert knowledge (Bagne et al. 2011;

Davison et al. 2012) and vulnerability indexes that combine impact models with an evaluation of species life-history traits and population status and distribution (Bertzky et al. 2011; Young et al. 2011; Comer et al. 2012). However, a study comparing three of these indexes has shown discrepant results concluding that they are very dependent on the factors considered and how they are calculated (Lankford et al. 2014).

Since the methodologies to assess species vulnerability to climate change start to multiply, it becomes ever more necessary to compare their relative success when applying to different groups of species and areas. Such analyses can contribute to future improvements and the development of innovations and novel approaches. To our knowledge, a comparative study of vulnerability assessment approaches with varying degrees of complexity has not been performed thus far. Methodologies that consider both potential abiotic impacts and species adaptive capacity are expected to produce improved (i.e. biologically more meaningful) results when compared with results obtained from the direct use of impact assessments, using, for example, bioclimatic models. The added advantage of such integrative, but more complex vulnerability assessment methodologies has to be evaluated since their results may have a direct consequence on the elaboration of effective and efficient climate change adaptation strategies.

The objective of this study is to compare and evaluate three different approaches, with increasing order of complexity and building on each other, for assessing species vulnerability to climate change as a part of a process to develop an adaptation strategy: (1) an impact assessment approach based on bioclimatic models (approach I), (2) an integrated vulnerability assessment approach which combines impact and adaptive capacity assessments (approach II) and (3) an approach that uses results of approaches I and II combined with an expert consultation workshop for validation of those results (approach III) (Fig. 2). These three approaches were tested on the Portuguese herpetofauna, a group identified as highly vulnerable to climate change (Araújo et al. 2006; Araújo et al. 2013).

Methods

Portuguese herpetofauna

The Portuguese terrestrial amphibiofauna includes 19 native species—12 anurans and 7 urodeles. Nine species (47 %) are Iberian endemics or near endemics (Rebelo et al. 2013). There are 28 native reptiles—16 saurians, 10 snakes and two freshwater turtles (Loureiro et al. 2008). Seven of these species (25 %) are Iberian endemics.

Among these 47 species, 13 are considered "endangered" (2), "vulnerable" (8) or "near-threatened" (3) according to the criteria of the IUCN red list (Cabral et al. 2005), and several others have not yet been assessed, but may in the future be classified as vulnerable (e.g. *Pelodytes ibericus*) (Rebelo et al. 2013). Factors presently threatening Portuguese herpetofauna include habitat change and fragmentation, invasive species, pollution and climate change (Loureiro et al. 2008; Rebelo et al. 2013).

Amphibians and reptiles are good candidates as indicators of climate change impacts because: (1) their present distribution in Portugal is well known (Loureiro et al. 2008); (2) temperature and moisture affect multiple aspects of their biology making many species vulnerable and some species extremely vulnerable to climate change direct impacts (Berry 2008; Henle et al. 2008); (3) they depend

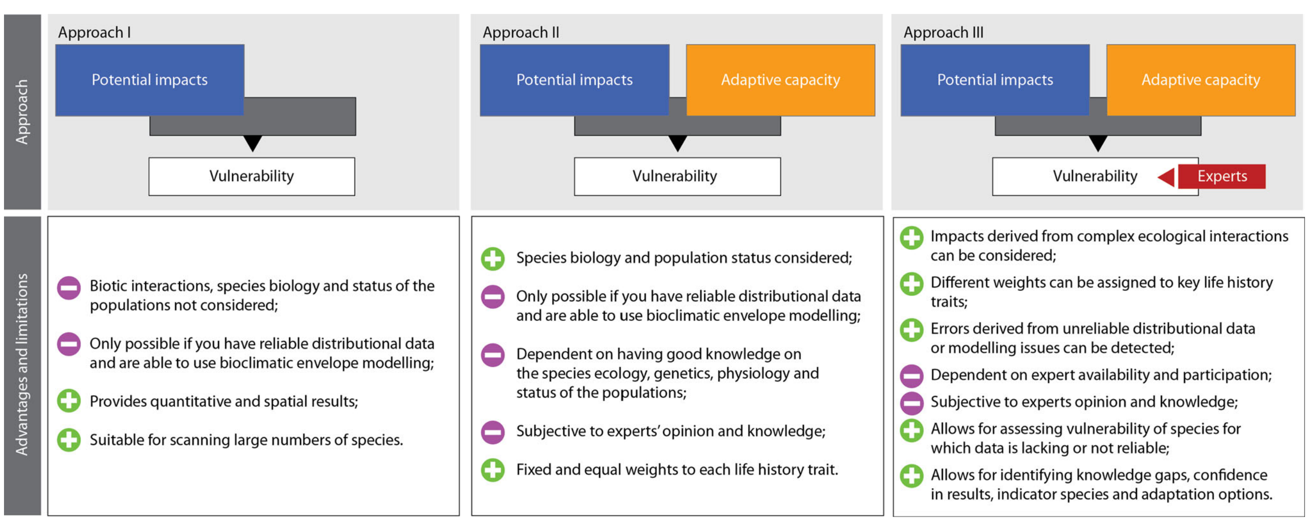

Fig. 2 Schematic representation of the three vulnerability assessment approaches analysed in this study (*top*—approaches), combined with the main advantages (*green* +) and limitations (*purple* −) of each approach (*bottom*—advantages and limitations)

on a large set of different habitats (from temporary ponds and streams to Mediterranean or Atlantic forests; (Loureiro et al. 2008); (4) their vulnerability to other stressors such as invasive species is already well understood (Rebelo et al. 2013), and interactions or synergies can thus be explored.

Current and future climate

Portugal's mainland most conditioning climate factors are latitude, orography and the effect of the Atlantic Ocean (Miranda et al. 2006). There is a large intra-annual temperature and precipitation variation: winters are characterized by high precipitation and low temperatures, while summers have low precipitation with high temperatures. North of Tejo River there is higher precipitation which is predominantly frontal and orographic, while the south is a semi-arid region with much lower precipitation, mainly associated with cyclogenetic activity (Trigo and DaCamara 2000; Costa et al. 2011). Significant changes in climate conditions for mid- and late century are projected for Portugal, including significant increase in temperatures, reduction in precipitation and increase in droughts (Miranda et al. 2006; Pulquério et al. 2014).

The two climatic zones have distinct flora and fauna compositions corresponding to different bioclimatic zones: the Lusitanian in the north and the Mediterranean in the south of Tejo River (Metzger et al. 2008). For the herpetofaunal communities, the north is occupied by species with Atlantic affinities, such as *Chioglossa lusitanica*, *Rana iberica*, *Podarcis bocagei* or *Vipera seoanei*, and the south occupied by species with Mediterranean affinities, such as *Pleurodeles waltl*, *Alytes cisternasii* or *Tarentola mauritanica*. The impacts of climate change are projected to shift the distribution of amphibian and reptile species to more northern latitudes; species already restricted to climate refugia in the north and in mountainous areas may become extinct (Araújo et al. 2012). The southern and drier areas are expected to lose a high proportion of species, while the north will lose some species and potentially gain others.

Comparison of vulnerability assessment approaches

Impact assessment approach

For the evaluation of the potential impacts of climate change on the Portuguese herpetofauna, we used the results of the bioclimatic envelope models produced by Araújo et al. (2012) for the Iberian Peninsula publicly available online (http://www.ibiochange.mncn.csic.es/iberiachange/). We compared the present potential distribution maps for Portugal with the ones of the period 2051–2080 considering the climate change scenario growth applied strategy (GRAS) (Fronzek et al. 2012). This scenario has been developed by FP6 project ALARM (http://www.alarmproject.net/) for large-scale application of dynamic ecosystem models and for biogeographical mapping and bioclimatic envelope modelling. GRAS is equivalent to the IPCC-developed scenario SRES A1F1.

For each species, we calculated two impact indexes: (1) the ratio impact index quantifying the percentage of change in potential distribution area between present and future; and (2) the overlap impact index quantifying the area within the intersection between future and present potential distributions divided by the present potential distribution (Bertzky et al. 2011). The ratio impact index varies between -100 and $+100$ and was categorized into five classes: very high (-100 to -70 %); high (-70 to -50 %); moderate (-50 to -30 %); low (-30 to -1 %); and no impact (0–100 %). The overlap impact index varies between 0 and 100 % and was categorized into four classes: very high (<30 %); high (30–50 %); moderate (50–70 %); and low (70–100 %). An average of these two indexes was calculated to obtain the expected impact, and species in the two highest vulnerability classes (i.e. high and very high) were considered as "vulnerable species". This assessment was not conducted for the four aquatic reptile species as there were no bioclimatic envelope models available.

Integrated vulnerability assessment approach

The methodology proposed by Bertzky et al. (2011) combines two indexes to produce vulnerability scores for each species: (a) a climate impact index (obtained from approach I) and (b) an adaptive capacity constrain index. In this study, the first index (a) was obtained directly from approach I.

The adaptive capacity constrain index (b) was calculated for each species as the sum of the individual scores for 11 different characteristics (Table 1). The scores should be attributed by experts and range from 0 (no constrain) to 2 (severe constrain) indicating how the characteristic is expected to constrain the species adaptive capacity (Bertzky et al. 2011). The index is then divided into three categories: low (<2); moderate (2–4); and high (>4). The authors' combined experience of many years working in climate change and Portuguese herpetofauna allowed to attribute these scores. The adaptive capacity index was produced for all Portuguese herpetofauna species, while the impact assessment could not be conducted for the four aquatic reptile species as there are no available data. The two indexes were combined into five classes (Table 2) according to the methodology of Bertzky et al. (2011). In order to compare the results with the other two approaches, species in the two highest vulnerability classes (i.e. critic and extremely critic) were considered as vulnerable.

The methodology described by Bertzky et al. (2011) indicates that the colonization restrictions should only be considered for species with <70 % overlap between present and projected climate. In this study, we assessed the colonization restrictions for every species because the herpetofauna generally has special habitat requirements, being distributed in small fragmented patches and having low dispersal ability between them. This means that dispersal ability is a limiting factor even for "normal" dispersion between breeding and non-breeding habitats and, therefore, colonization ability can be limiting to species adaptability even when there is a large overlap between present and future potential distributions.

Integrated vulnerability assessment and validation approach

This approach builds on the previous two approaches, integrating expert opinion for validation or rebuttal of its results. We conducted a one-day workshop with nine independent experts on Portuguese herpetofauna physiology, ecology and genetics and two technicians from the Portuguese Institute for Nature Conservation and Forests (ICNF), responsible for developing the National Climate Change Adaptation Strategy (NCCAS) for Biodiversity. In this workshop, we presented the vulnerability assessment results obtained through approaches I and II. The experts were asked to review these classifications, to identify any disagreements with the results, to indicate the reasons for disagreeing and to propose a final consensual list of vulnerable species. The workshop also allowed for assessing the vulnerability of the four reptile species for which we did not have impact models and were therefore not assessed with approaches I and II. These four species were classified as vulnerable or not vulnerable to climate change based on the results of the adaptive capacity constrain index and expert judgement.

Table 1 Characteristics used to assess species adaptive capacity constraints—approach II

General restrictions	
1	Small population and/or distribution in Portugal
2	Low survival and/or productivity rates
3	Long generation times/long life cycles
4	Declining populations in Portugal
5	Low genetic diversity
6	Specialized and uncommon habitat requirements
7	Narrow niche
8	Critical association with other vulnerable species
Colonization restrictions	
9	Barriers to dispersal (water, topography, man-made)
10	Limited dispersal and/or colonization capacity
11	Mainly distributed in fragmented habitats

Adapted from Bertzky et al. (2011)

Results

Overall, all three approaches indicate considerably high percentages of vulnerable species, identifying more amphibians than reptiles as vulnerable to climate change (Fig. 3). The three approaches agree in their vulnerability score for 35 out of the 43 species studied, the largest differences between approaches being found between approach III and the other two approaches. Approach I shows the highest number of vulnerable species of all three approaches.

In the impact assessment approach (approach I), 11 amphibian (58 %) and 11 reptile (46 %) species were classified as vulnerable to climate change (Fig. 3), while ten amphibian (53 %) and nine reptile (38 %) species were considered vulnerable according to the integrated vulnerability assessment approach (approach II). Approach II changed the vulnerability score obtained in approach I for three species from vulnerable to not vulnerable (Table 3), all of which had a classification of moderate impacts in approach I (impact = 2).

In the integrated vulnerability assessment and validation approach (approach III), experts disagreed with results of approaches I and/or II for 8 of the 43 species assessed (19 %). Disagreements with approach I were found for two species, with approach II for one species and with both approaches I and II for five species (Table 3). From these

Table 2 Vulnerability categories, combining species adaptive capacity constraints and expected climate impacts used in approach II

		Climate impact category			
		Low	Moderate	High	Very high
Adaptive capacity constraint	Low	Low	Moderate	High	Very high
	Moderate	Moderate	High	Very high	Critical
	High	High	Very high	Critical	Extremely critical

Adapted from Bertzky et al. (2011)

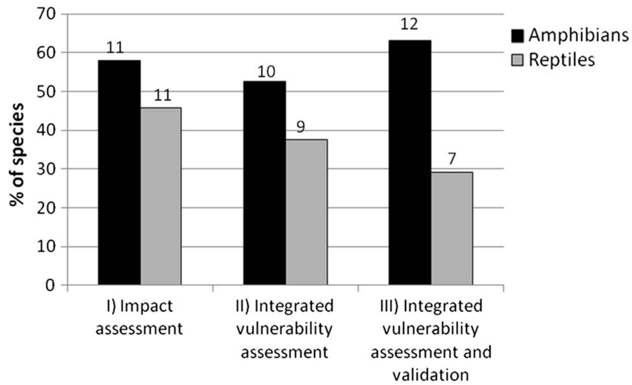

Fig. 3 Percentage of amphibian and reptile species classified as vulnerable using the three selected approaches. Numbers on the *top* of each *bar* represent the number of vulnerable species. Total number of species assessed was 19 amphibians and 24 reptiles

eight species, three were considered vulnerable by the experts and five as not vulnerable.

Reasons identified by the experts for the proposed changes to results of approaches I and II were related to the bioclimatic models for all but one species—*Pelobates cultripes*. For this species, it was considered that the bioclimatic models were correctly showing high direct impacts. However, the adaptive capacity analysis was considered incorrect by the experts, as *P. cultripes* should be classified as vulnerable considering its high dependence on temporary water bodies, which will be significantly affected by climate change.

Main consensual reasons referred by experts to disagree with the results of the bioclimatic envelope models were:

1. Problems with data on species distributions. For many of the species under study, the data on potential present distribution were considerably different to their actual distribution; for eight species (19 %), the present distribution was overestimated, and for one species (2 %), there was an underestimation.
2. Problems with the model predictions. Experts indicated that results of the bioclimatic envelope models were unreliable for at least seven species (16 %) as other works have been suggesting quite different potential impacts (Carvalho 2010).
3. The fact that indirect impacts are not evaluated in the bioclimatic envelope models used. Specifically, the association of species with very vulnerable habitats (e.g. temporary ponds) has been pointed by experts as very relevant for defining species vulnerability to climate change.

The experts also indicated that for two species not ranked as vulnerable (*Salamandra salamandra* and *Podarcis hispanica*), a reassessment of their vulnerability should be carried out when new data on the distribution of the different genetic groups these species are composed of—potentially reacting differently to climate change—are available.

As for the vulnerability of the four reptile species for which impact models did not exist, the experts considered three of them vulnerable, essentially due to their very high dependence on water bodies that will be affected by climate change.

In total, 12 amphibians and 7 (+3) reptiles (Fig. 3) were classified as vulnerable to climate change in the integrated vulnerability assessment and validation approach (approach III).

Discussion

Appraisal of the vulnerability assessment approaches

Our study compares the three types of approaches most commonly used to assess species vulnerability to climate change (Bertzky et al. 2011; Carvalho et al. 2011; Davison et al. 2012) and shows disagreements between the three approaches for approximately a fifth of the species studied (8 of the 43 species).

Based on our results, we argue that: (1) all approaches present limitations; (2) results obtained with approach II are not meaningfully different from results obtained with approach I; (3) approach III can increase confidence in the results of any of the previous approaches, provide a "best informed guess" based on available knowledge and provide a good basis for action in spite of uncertainty; and (4) the choice of a methodology will ultimately depend on the purposes of the vulnerability assessment.

Approach I presents the limitations associated with the use of bioclimatic envelope models which have been extensively reviewed and include: limited availability or poor quality of baseline data to be used in the models; limitations of the modelling methodologies used; and lack of incorporation of possible indirect impacts on the distribution of a species, such as the reduction in species habitats (Guisan and Thuiller 2005; Jeschke and Strayer 2008; Thuiller et al. 2008). These limitations have been highlighted in the validation workshop where experts disagreed with results from this approach for seven species. The bioclimatic envelop models provide only a quantitative assessment of the climate change impact by providing a measure of species sensitivity to future abiotic conditions. Thus, they do not provide a true vulnerability index, and interpretation of the results should be done with caution (Araújo and Peterson 2012).

Approach II, although being a more integrated approach, inherited the limitations of its components. As seen in this

Table 3 List of species and vulnerability scores obtained with the different approaches and reasons given by experts to change the vulnerability score

	Species	I	II	III	Disagreement reason
Amphibians	*Chioglossa lusitanica*	**Y**	**Y**	**Y**	
	Pleurodeles waltl	N	N	N	
	Salamandra salamandra	N	N	N	
	Triturus boscai	N	N	N	
	Triturus helveticus	**Y**	**Y**	**Y**	
	Triturus pygmaeus	N	N	**Y**	Baseline data and indirect impacts
	Triturus marmoratus	**Y**	**Y**	**Y**	
	Alytes cisternasii	**Y**	**Y**	**Y**	
	Alytes obstetricans	**Y**	**Y**	**Y**	
	Discoglossus galganoi	**Y**	**Y**	**Y**	
	Pelobates cultripes	**Y**	N	**Y**	Weighting of indirect impacts
	Pelodytes ibericus	N	N	**Y**	Modelling and indirect impacts
	Pelodytes punctatus	**Y**	**Y**	**Y**	
	Bufo bufo	N	N	N	
	Bufo calamita	**Y**	**Y**	**Y**	
	Hyla arborea	**Y**	**Y**	N	Baseline data
	Hyla meridionalis	N	N	N	
	Rana iberica	**Y**	**Y**	**Y**	
	Pelophylax perezi	N	N	N	
Reptiles	*Hemidactylus turcicus*	N	N	N	
	Tarentola mauritanica	N	N	N	
	Chamaeleo chamaeleon	N	N	N	
	Anguis fragillis	**Y**	**Y**	**Y**	
	Blanus cinereus	N	N	N	
	Acanthodactylus erythrurus	N	N	N	
	Timon lepidus	N	N	N	
	Lacerta schreiberi	**Y**	**Y**	**Y**	
	Iberolacerta monticola	Y	Y	Y	
	Podarcis bocagei	**Y**	N	N	Modelling
	Podarcis carbonelli	**Y**	**Y**	**Y**	
	Podarcis hispanica	N	N	N	
	Psammodromus algirus	N	N	N	
	Psammodromus hispanicus	**Y**	**Y**	N	Baseline data
	Chalcides bedriagai	**Y**	**Y**	N	Modelling
	Chalcides striatus	N	N	N	
	Hemorrhois hippocrepis	N	N	N	
	Coronella austriaca	**Y**	**Y**	**Y**	
	Coronella girondica	**Y**	N	N	Indirect impacts
	Rhinechis scalaris	N	N	N	
	Macroprotodon cucullatus	N	N	N	
	Malpolon monspessulanus	N	N	N	
	Vipera latastei	**Y**	**Y**	**Y**	
	Vipera seoanei	**Y**	**Y**	**Y**	
	Natrix maura	a	a	N	
	Natrix natrix	a	a	**Y**	
	Emys orbicularis	a	a	**Y**	
	Mauremys leprosa	a	a	**Y**	

I—impact assessment approach; II—integrated vulnerability assessment approach; III—integrated vulnerability assessment and validation approach (scores are Y (bold) = species with high vulnerability and N (unbold) = species with low vulnerability)

[a] No data available

study, the results of this approach were very dependent on the results of the bioclimatic modelling and therefore inherited its limitations, especially those relating to modelling errors and unreliable or missing data. Experts disagreed with the results of this approach for six species. This means that according to experts, approach II only improved the results of approach I for two species (5 %). On the other hand, for another species, *P. cultripes*, this approach changed the result of approach I from vulnerable to not vulnerable, whereas the experts (approach III) did not agree with this change.

These results seem to indicate that approach II, although allowing for the integration of factors contributing to the species adaptive capacity, such as indirect impacts, species life-history traits and population status, does not add much information to bioclimatic models' results (approach I). This is, at least in part, due to the way the index was computed: approach II is only capable of changing the overall vulnerability class for species classified with moderate impacts using the bioclimatic envelope models. Changing the weights of the different components of approach II could therefore lead to different results. In fact, according to the experts (approach III), more weight should be given to adaptive capacity constraint if, for example, there is a total dependence on a habitat that is under high threat from climate change, as was the case for *P. cultripes*. The reasoning is that if a species cannot survive without a particular habitat and the habitat is highly vulnerable, this factor alone should be enough to rank that species as highly vulnerable to climate change. Young et al. (2014), when evaluating studies that apply their vulnerability assessment index, also referred to the same type of issue: for species dependent on vulnerable habitats such as wetlands, their index yielded what users have considered lower than expected vulnerability scores.

The development of methodologies for comprehensive vulnerability assessments that consider both bioclimatic envelope models and species traits is rapidly growing (Bagne et al. 2011; Comer et al. 2012; Davison et al. 2012; Lankford et al. 2014). However, according to (Lankford et al. 2014), results obtained with different indexes can greatly vary, depending on the specific factors considered and even on the way the questions are worded. Further, all these indexes seem to face this same problem: weighting of different factors is fixed, regardless that according to experts, it should depend on the species in question. Different factors may be limiting for different species or groups: some species may be highly dependent on specific habitats or food type, and their vulnerability will therefore reflect such dependence. One way to deal with this issue is to use a weighted product model or a logic model where, for example, instead of averaging or summing all scores, the number of high scores (in this case, the number of factors that significantly reduce a species adaptive capacity) is calculated.

On the other hand, bioclimatic envelope models are becoming more sophisticated and realistic (Heikkinen et al. 2007; Thuiller et al. 2008; Brook et al. 2009) and will increasingly allow for more integrated approaches. These models might thus, in the future, include impacts on the species habitats and trophic relations or information on species biological traits and other interacting pressures, potentially increasing our ability to quantify species' vulnerability to climate change.

The choice of the most appropriate methodology for a specific area, habitat or group of species will depend on available data, the number of species to be assessed and the availability of experts to contribute to the different phases of the assessment (e.g. to the production of adaptive capacity constrain indexes and validation workshops). However, the ultimate goal of such an assessment may be the most important factor to have in mind, as the different approaches tested in this study can be used for different purposes.

Approach I has proven efficient to capture the broad patterns of herpetofauna vulnerability to climate change. For example, this approach results agreed with those of the other approaches in identifying the most vulnerable groups (amphibians more vulnerable than reptiles; species associated with temporary water bodies more vulnerable than others) and the most vulnerable areas (north and mountainous areas more vulnerable than others). Therefore, this approach seems to be a good choice for screening large numbers of species and identify broad vulnerability patterns, as they are relatively easy to apply to such numbers, assuming that distribution data are available. Despite the limitations and many problems already identified of bioclimatic envelope models (Guisan and Thuiller 2005; Jeschke and Strayer 2008; Araújo and Peterson 2012), they do provide quantitative and spatial data that are useful when assessing climate change impacts in biodiversity.

Considering the small differences obtained between approaches I and II, one can conclude that the impact assessment approach would have provided a useful first approximation as to the potential impact of climate change on biodiversity. Therefore, usefulness of approach II seems quite limited, especially if there is the possibility to have a validation workshop with experts, as the experts will bring into the table the adaptive capacity constraint analyses. However, for species with unknown or incomplete distribution data, and therefore for which the production of bioclimatic envelope models is less reliable, using an adaptive capacity constrain index can lead to informative results on differential vulnerability to climate change. Furthermore, for developing adaptation strategies, it is useful to understand which key factors are limiting species adaptive capacity. If this is one goal of the assessment, approach II may prove more useful than an impact assessment approach.

Approach III can be useful when a detailed assessment of species vulnerabilities to climate change is needed. According to our results, approach III seems to have some advantages. First, this approach allows for integration of results obtained with different approaches together with expert knowledge from different fields (e.g. genetics or conservation) to provide a more robust analyses of species vulnerability. Second, it allows for a better understanding of knowledge gaps and the uncertainties involved in the vulnerability assessment, regardless of the approach chosen. Third, it is appropriate in situations where data availability and quality are reduced and/or poor and are not dependent on the availability of bioclimatic envelope models. Finally, such an approach allows for collecting other types of results such as identifying key attributes leading to vulnerability, indicator species and defining informed and specific adaptation measures and strategies.

However, it should be noted that the use of expert consultation is not without problems: it is dependent on experts' availability; large number of species add complexity and time to the consultations; there is the need to have ecological, physiological and genetics knowledge of the species considered; and any expert judgment is subjective.

Our results seem to support the observations that the potential effects of climate change on biodiversity are complex and difficult to predict (Williams et al. 2008; Bagne et al. 2011; Lankford et al. 2014). Considering the available methodologies at the moment, we cannot yet fully forecast the combined effects that will result from direct and indirect impacts which depend on species phenological, physiological and evolutionary responses (Williams et al. 2008; Bellard et al. 2012; Staudinger et al. 2012; Lankford et al. 2014). The use of multiple methods can provide better insight on uncertainties for different species or groups of species and may help to identify species for which methods disagree. Studies to further compare these and other methodologies for a range of group species and regions will be fundamental to understand their usefulness and applicability. It would be interesting to compare other of the many new vulnerability indexes being developed recently (Bagne et al. 2011; Comer et al. 2012; Davison et al. 2012) with bioclimatic envelope models to see whether our conclusions would still hold.

Contributions towards a Portuguese herpetofauna vulnerability assessment and adaptation strategy

Overall, 63 % of the amphibians and 36 % of the reptiles were classified as vulnerable to climate change. Iberian Peninsula endemics were almost completely included in the list (seven out of the nine amphibians and five out of the seven reptiles). Another common pattern was the indication of vulnerability for almost all the species that are currently mainly found in the north-western Mountains—five out of six amphibians and six out of seven reptiles.

Our results, clearly indicating that amphibians are more vulnerable than reptiles, are in line with most impact assessments done for other geographical regions (e.g. Araújo et al. 2006; Henle et al. 2008; Salice 2012; European Union 2013) and reflect the fact that reptiles are in general better adapted to dry environments, the exceptions being the species associated with freshwater habitats such as the turtles (*Emys orbicularis* and *Mauremys leprosa*) or the water lizard (*Lacerta schreiberi*) or species already limited to mountainous areas (e.g. *Iberolacerta monticola*).

The factor that most frequently contributed to the classification of an amphibian as vulnerable was its dependence on temporary ponds to reproduce. Other anthropogenic pressures have been shown to produce the same pattern among Iberian amphibians: species associated with temporary ponds have been shown to be the most vulnerable to the introduction of exotic predators (Cruz et al. 2006) and to land use changes (Ferreira and Beja 2013). Amphibian vulnerability to climate change in this area seems to follow this same pattern, being mainly a result of the vulnerability of temporary ponds to changing temperature and precipitation regimes.

For both groups, most of the species that are already limited to mountainous areas in the north-west of Portugal (e.g. *C. lusitanica*, *I. monticola*, *V. seoanei*) were considered vulnerable, regardless of the approach followed. This is because the Atlantic climate that characterizes the mountainous north-west covers already a relatively small part of the country, and most of the species that are dependent of this climate are probably retreating under an advancing Mediterranean-type climate since the end of the last glaciation (Loureiro et al. 2008).

As expected, rarity *per se* was not a good predictor of vulnerability to climate change, as some species that are currently rare but restricted to the semi-arid south-west (e.g. *Hemidactylus turcicus* or *Chamaeleo chamaeleon*) were not considered vulnerable by any of the approaches. Furthermore, high vulnerability indexes were attributed to species that are presently common and apparently maintain stable populations (e.g. *Discoglossus galganoi*, *Natrix natrix*). This means that climate change may pose new challenges for biodiversity conservation and that a re-evaluation of species conservation status should be conducted shortly having this vulnerability assessment results in consideration.

Seven species were identified as indicator species in the workshop considering not only their vulnerability to climate change, but also other factors such as (1) being associated with vulnerable habitats, (2) being subjected to other pressures that can have synergistic effects with climate change (namely competition from invasive species), (3) being potentially favoured by the expected climate changes, (4) the existence of baseline information on the populations and (5) the monitoring efforts required.

Based on the results of the vulnerability assessment, 29 measures and actions were identified and prioritized during the expert workshop and have been included in NCCAS (Araújo et al. 2013). The use of a workshop has allowed to: (1) identify vulnerable species that will be subjected to periodic revision of their conservation status, (2) identify indicator species that will be monitored, (3) identify geographic regions that will suffer higher biodiversity loss, (4) prioritize measures and actions for herpetofauna conservation and (5) identify knowledge gaps and define methodologies to overcome them. With the expected climate changes, improving methodologies for vulnerability assessment such as the ones tested here will be critical in supporting the development of effective adaptation strategies.

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