# Waterbody availability and use by amphibian communities in a rural landscape

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Rural landscapes in central and eastern Europe provide valuable ecosystem services and support high levels of biodiversity. These landscapes face an increasing pressure from human development and changes in agricultural practices. Pond-breeding amphibians and their breeding habitats are especially vulnerable to land-use changes. We studied waterbody use by amphibians in a rural landscape from Haţeg Geopark, Central Romania, a region where large areas are still under traditional land use. We surveyed 55 waterbodies, characterized them and their surrounding terrestrial habitats with 22 variables. Amphibians were more sensitive to waterbody-related variables than to landscape parameters. Man-made waterbodies had lower species richness than natural ones, but often represent the only breeding habitats available. The low importance of the landscape variables for amphibians is the result of traditional and environment-friendly land management, which maintains a mosaic landscape where the optimal terrestrial habitats for amphibians are still well represented.

*Key words:* conservation, man-made waterbodies, temporary pond, traditional land use

## INTRODUCTION

Rural landscapes of central and eastern Europe (CEE) are still characterized by predominantly traditional agricultural practices, creating and maintaining landscape mosaics with high species richness (Bignal & McCracken, 2000; Palang et al., 2006). These diverse rural landscapes are now under threat by land-use intensification, partly encouraged by the Common Agricultural Policy following the European Union expansion (Young et al., 2007), land abandonment or  land-use change (Plieninger et al., 2006; Kuemmerle et al., 2009).

Amphibians are declining worldwide and are considered the world's most endangered vertebrates (Baillie et al., 2004; Stuart et al., 2004). Pond-breeding amphibians are particularly sensitive to habitat changes, due to their complex life-cycle and the need of interconnected aquatic and terrestrial habitats for breeding, feeding and overwintering (Moran, 1994; Wells, 2007). Amphibians often persist in human-altered environments, because human activity indirectly creates and maintain suitable habitats for them (Hazell et al., 2004; Herzon & Helenius, 2008; Hartel et al., 2010a, b). Artificial waterbodies created for various purposes may act as important breeding habitats for amphibians (Beebee, 1997; Knutson et al., 2004; Dalbeck & Weinberg, 2009, Curado et al., 2011), and often support high amphibian species richness (Beebee, 1997; Abellan et al., 2006; Ruggiero et al., 2008; Thiere et al., 2009). Documenting amphibian habitat use in human-modified landscapes is important for promoting efficient conservation and management measures.

In many regions of Romania, rural landscapes are still managed in a traditional or environmentally friendly way. After 1990, the use of pesticides and chemical fertilizers decreased steadily (Ciaian & Pokrivcak, 2007; Turnok, 1996). Studies carried out in lowland areas of Romania show that traditional agricultural practices are crucial for the creation and maintenance of self-sustainable and species-rich amphibian communities (Hartel et al., 2010b). Our present study aims to i) assess the use of waterbodies by amphibians in a traditional rural landscape, ii) identify waterbody characteristics useful for predicting the amphibian species occurrences, and iii) evaluate the importance of man-made habitats for the persistence of amphibians.

## MATERIALS AND METHODS

### Study area

Haţeg Geopark (HG) is located in the western part of Romania, in the Haţeg depression (N 45°12'–45°18', E 22°25'–23°20') with an area of 1024 km$^2$ (Fig. 1). HG has the highest human population density for a protected area in Romania (34.5 inhabitants/km$^2$). The landscape is highly diverse and traditional agricultural practices are still predominant. Forests prevail (48%), followed by pastures and natural grassland (19%), and arable land (18%). Only 4% of the area is built-up. The mean annual temperature varies between 6 and 8° C, and the mean annual rainfall varies between 700 and 800 mm/year.

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**Waterbody survey**

Waterbodies were surveyed during the period 2004-2008. Waterbodies were visited multiple times during and after the breeding season of amphibians in order to detect breeding adults, eggs, larvae and/or juveniles. Due to the predominant temporary and small-sized character of the ponds, we were confident that we detected each inhabiting amphibian species. The geographic coordinates and altitude were measured with a handheld Garmin GPS device. Water temperature, pH and conductivity were measured with a portable Oakton Waterproof pH/mV/C Meter.

Each waterbody was characterized by the following variables: origin (natural or man-made), type (puddle, wetland, drainage ditch, pond, reservoir), risk of desiccation (low, medium, high), maximum water depth (cm), area ($m^2$), water vegetation (presence or absence of macrophytes), presence of fish and invertebrate predators, temperature ($^oC$), pH and conductivity (μS/cm). The presence of leeches, aquatic Coleoptera, Odonata larvae and Heteroptera was used to evaluate invertebrate predation risk and was assessed by dip-netting (Knutson et al., 2004). The presence/absence of fish was based on visual assessments, dip-netting and information provided by local fishermen. We considered a pond complex if at least two other water bodies were present in a 50 m radius. Waterbodies where eggs, larvae and/or juveniles were found were considered breeding habitats. The presence of paved roads was considered (if the distance between waterbodies and roads was less than 50 m), since they present a high mortality risk for amphibians due to intense traffic (Hels & Buchwald, 2001). We combined the green frogs *Pelophylax esculentus* and *P. ridibundus* due to difficulties of identification in the field (Pagano & Joly, 1998). *Pelophylax lessonae* could be distinguished from the other green frogs.

**GIS analysis**

Landscape variables were obtained from the Corine Land Cover 2000 (CLC2000) seamless vector database, finalized in the early 1990s as part of the European Commission programme CORINE (European Environment Agency, 2009). The CLC2000 interpretation for our study area in Romania is based on year 2000 satellite imagery. The land cover details are limited to a minimum mapping unit (MMU) of 25 ha, and only features that are relatively stable in time are represented (Heymann et al., 1994).

The land cover was extracted for each waterbody location in a 500 m radius (an area of 78.5 ha) using the Buffer analysis tool in ArcGIS Desktop 9.3.1 ArcInfo license (ESRI, 1995-2010). Each buffer area was intersected with the CLC2000 vector layer in order to generate an independent set of polygons that represented the land cover. A query in Microsoft Access 2007 was used to aggregate the results and generate the percentages of each land cover within each buffered area around waterbodies. For the 14 resulting CLC classes, categories with less than two occurrences were included in the nearest CORINE level (Table 1).

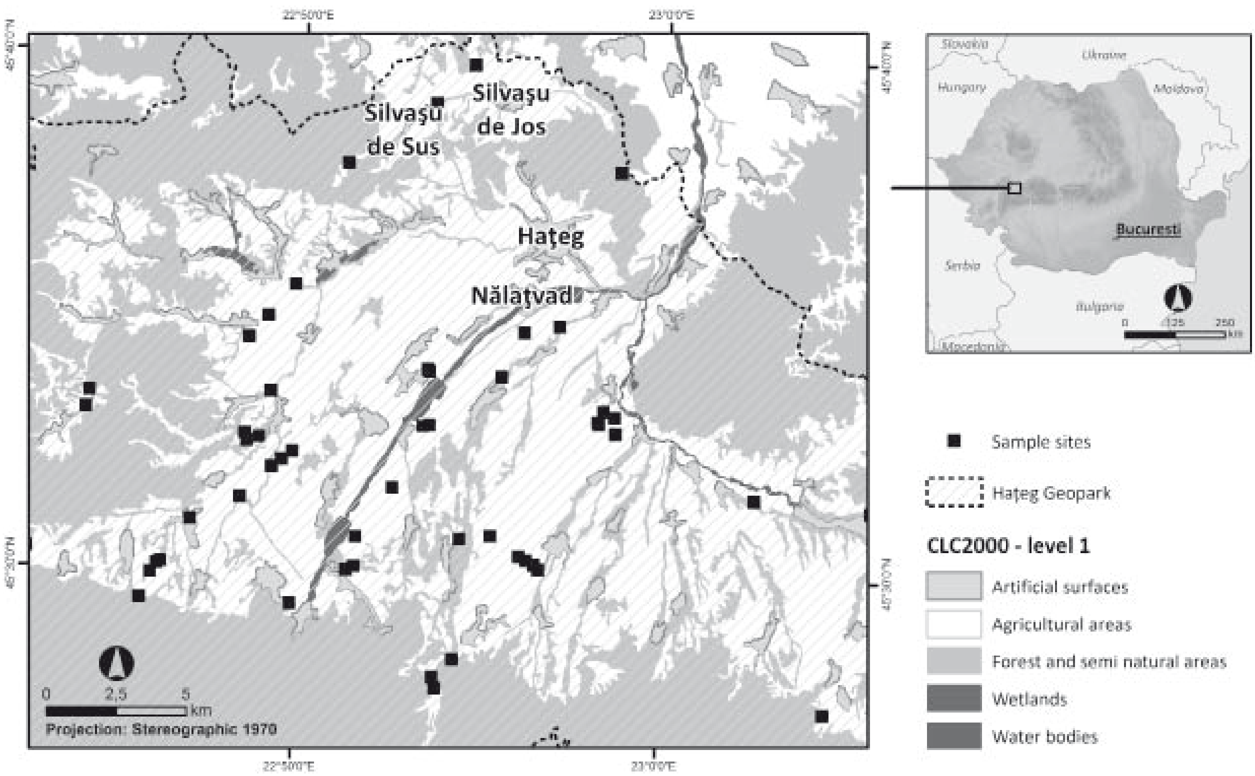


**Fig. 1.** Waterbodies inventoried in the Haţeg Geopark (black squares) in Romania (bottom left). Data sources: CLC2000 (Copyright EEA, Copenhagen, 2007), Geo-spatial.org, Ministry of Environment and Forests, Romania.

**Table 1.** Corine Land Cover codes and their assigned variables.

| CORINE code | Variables |
|---|---|
| 1.1.2.and 1.2.1. | Settlement |
| 2.1.1. | Arable land |
| 2.2.2. | Orchard |
| 2.3.1. and 3.2.1. | Pastures |
| 2.4.2. | Complex cultivation |
| 2.4.3. | Arable land with areas of natural vegetation |
| 3.1.1. and 3.1.2. | Forest |
| 3.2.4. | Transitional woodland-shrub |
| 5.1.2., 4.1.1. and 5.1.1. | Inland water |

**Data analysis**

We considered that a given species of amphibian was present in a particular waterbody if at least one life stage was detected. All continuous variables were *z*-transformed (standardized to an average of zero and a standard deviation of one) to increase the comparability of predictors (Ćirović et al., 2008), and a Hellinger transformation was applied to categorical variables to meet the assumption of Principal Component Analysis (PCA). PCA was used to remove intercorrelations inherently present within the explanatory variables, and to reduce the number of predictors. Eight principal components (Table 2) were considered significant under a broken-stick distribution (Jackson, 1993; Diniz-Filho et al., 1998). The VARIMAX method with Kaiser normalisation was used as a rotation method. The first principal component (Comp.1) was represented by water chemistry (Wch), the second (Comp.2) by water volume (Wv), the third (Comp.3) by rural landscape (Rl), the fourth (Comp.4) by high impacted adjacent land use (Hilu), the fifth (Comp.5) by the woodlands (Wl), the sixth (Comp.6) by inland water, the seventh (Comp.7) by arable lands with areas of natural vegetation, and the eighth (Comp.8) by complex cultivation (for details see the coefficients of the first eigenvalues in Table 2). The eight variables extracted by PCA axes were used to build 12 *a priori* models (Table 3).

An information-theoretic approach was used to identify the appropriate models for predicting the occurrence of individual amphibian species (Generalized Linear Model (GLM) assuming binomial error and a logit link function) and species richness (GLM assuming normal errors and a identity link function; the assumptions required by GLM were checked prior to the analysis). In order to reduce the heteroscedasticity of amphibian species richness (dependent variable), a $\log_{10}$ transformation was applied. The species that occurred in less than five sites were excluded from the single species modeling. For each model, the AIC value

**Table 2.** PCA loadings of explanatory variables.

| Explanatory variables | | Comp.1 | Comp.2 | Comp.3 | Comp.4 | Comp.5 | Comp.6 | Comp.7 | Comp.8 |
|---|---|---|---|---|---|---|---|---|---|
| Habitat origin | Ho | 0.002 | -0.005 | -0.007 | 0.003 | -0.001 | 0.000 | 0.000 | 0.004 |
| Habitat type | Ht | 0.000 | 0.002 | 0.003 | -0.001 | 0.001 | 0.000 | 0.000 | -0.002 |
| Desiccation risk | Dr | -0.048 | -0.220 | -0.067 | 0.010 | 0.049 | -0.024 | -0.083 | 0.040 |
| Surface area | S | 0.029 | 0.375 | 0.145 | -0.042 | -0.119 | -0.008 | 0.153 | 0.160 |
| Maximum depth | Md | -0.113 | 0.572 | 0.071 | -0.079 | 0.197 | 0.082 | -0.020 | -0.165 |
| Water temperature | Wt | 0.357 | -0.075 | -0.003 | 0.025 | -0.029 | 0.063 | -0.067 | 0.008 |
| pH | - | 0.360 | -0.046 | 0.018 | 0.019 | -0.031 | 0.072 | -0.036 | 0.042 |
| Conductivity | C | 0.345 | -0.055 | -0.053 | -0.068 | 0.104 | -0.084 | 0.114 | 0.086 |
| Arable land | Al | -0.029 | 0.074 | -0.102 | -0.531 | 0.055 | 0.117 | -0.190 | -0.186 |
| Arable land with areas of natural vegetation | Alnv | -0.006 | 0.077 | -0.016 | 0.017 | 0.037 | -0.006 | 0.780 | -0.056 |
| Complex cultivation | Cc | 0.045 | 0.029 | -0.122 | -0.005 | 0.100 | 0.043 | 0.016 | 0.893 |
| Transitional woodland shrubs | Tws | 0.030 | 0.115 | 0.013 | 0.053 | 0.587 | -0.152 | -0.014 | 0.095 |
| Pasture | Past | -0.074 | -0.009 | -0.263 | 0.581 | 0.217 | 0.157 | -0.206 | -0.201 |
| Settlement | St | 0.098 | -0.206 | 0.605 | 0.286 | -0.166 | 0.284 | 0.132 | 0.110 |
| Forest | Fst | 0.008 | -0.086 | 0.012 | -0.056 | -0.514 | -0.393 | -0.169 | -0.013 |
| Orchard | Ft | 0.056 | 0.006 | 0.466 | -0.196 | 0.166 | -0.244 | -0.300 | -0.047 |
| Inland water | Iw | 0.034 | 0.097 | 0.048 | 0.045 | -0.032 | 0.655 | 0.015 | 0.065 |
| Pond complex | Pc | 0.023 | -0.034 | -0.046 | 0.015 | 0.009 | -0.004 | 0.007 | 0.005 |
| Vegetation | Veg | 0.004 | 0.063 | 0.078 | -0.009 | -0.006 | 0.003 | 0.044 | 0.013 |
| Predators | Pr | 0.035 | 0.022 | 0.011 | 0.045 | -0.010 | 0.037 | 0.076 | 0.054 |
| Fish | Pf | -0.005 | 0.022 | -0.001 | -0.002 | 0.014 | 0.010 | 0.001 | 0.005 |
| Roads | Rd | 0.007 | 0.001 | -0.048 | -0.024 | -0.003 | -0.009 | 0.027 | -0.053 |
| Percentage of explained variation | | 16.236 | 28.124 | 37.839 | 46.104 | 53.282 | 59.590 | 65.174 | 70.137 |

**Table 3.** Candidate models used in the analysis of species richness and occurrence of individual amphibian species in ponds from HG. Wch - water chemistry, Wv - water volume, Rl - rural landscape, Hilu - high impacted land use, Wl - woodland, Iw - inland waters, Alveg - arable land with areas of natural vegetation, Cc - complex cultivation

| | Model name | Covariates |
|---|---|---|
| Model 1 | Global | Wv, Wch, Rl, Hilu, Wl, Iw, Alveg, Cc |
| Model 2 | Abiotic | Wv, Wch |
| Model 3 | Water chemistry | Wch |
| Model 4 | Water volume | Wv |
| Model 5 | Adjacent land use | Rl, Hilu, Wl, Iw, Alveg, Cc |
| Model 6 | Adjacent human land use | Rl, Hilu, Alveg, Cc |
| Model 7 | Cropland | Alveg, Cc |
| Model 8 | Woodland | Wl |
| Model 9 | High impacted land use | Hilu |
| Model 10 | Low impacted land use | Alveg |
| Model 11 | Inland waters | Iw |
| Model 12 | Rural landscape | Rl |

was calculated using correction for small sample size ($AIC_c$) (Burnham & Anderson, 2002). The models were then ranked according to their $AIC_c$ values, the best model having the smallest $AIC_c$ value. Delta ($\Delta$) $AIC_c$ was computed as the difference between each model and the best model. The Akaike weights ($w_i$) express the weight of evidence favouring the model as the best of all models. Akaike weights were used to calculate parameter estimates and their variances using "model averaging". Statistical procedures were implemented in R 2.1.0 (R Development Core team, 2005). The model selection and model averaging were performed with the AICcmodavg package (Mazerolle, 2009).

## RESULTS

We identified 55 waterbodies in the study area (Fig. 1). Most of them were small and temporary (82% less than 50 $m^2$ in area, Table 4). Man-made habitats were slightly prevalent (56%). Amphibians were present in 95% of waterbodies, but only 66% of the sites were used for breeding. Thirty-five percent of breeding habitats were of anthropogenic origin, and 31% were natural (Fig. 3). We recorded ten amphibian species and a species complex: *Bombina variegata*, *Rana dalmatina*, *R. temporaria*, *Hyla arborea*, *Bufo bufo*, *Pseudepidalea viridis*, *Triturus cristatus*, *Lissotriton vulgaris*, *Salamandra salamandra, Pelophylax lessonae* and *P. esculentus* complex. *Pseudepidalea viridis*, *S. salamandra*, *P. lessonae* and *R. temporaria* were present in less than five sites. The most common species was *B. variegata,* with occupancy of 92% in natural and 81% in man-made habitats, followed by *R. dalmatina* and *P. esculentus* complex with 29% and 26%, respectively (Fig. 2A). Species richness was significantly higher in natural habitats than in man-made habitats ($F$=5.773, $P$=0.02) (Fig. 2B). *Bombina variegata*, *R. dalmatina* and *P. esculentus* complex used a high percentage of sites for reproduction. *Bufo bufo* offspring was found only in man-made habitats (Fig. 3).

The eight variables extracted by PCA explained more the 70% of the original variation (Table 2). Table 5 summarizes the model selection results and Table 6 provides parameter estimates of the used covariates. The best model for species richness was Water Volume, followed by the Abiotic and Woodland model (Table 5). Water Volume was also the best model for the occurrence of *B. variegata*, *R. dalmatina*, *T. cristatus* and *L. vulgaris*. The Abiotic and Woodland models best predicted the occurrences of *B. bufo* and *R. temporaria*, respectively. The best model to predict the presence of *Hyla arborea* was Rural Landscape, whereas the presence of *P. esculentus* complex was best predicted by Inland Water. The model-averaged parameter estimate (Table 6) indicated that water volume was the best predictor for species richness, and the occurrence of *R. dalmatina*, *T. cristatus* and *L. vulgaris*. Woodland was negative for *R. temporaria*, and Inland Water was positive for *P. esculentus* complex.

**Table 4.** Altitude, physical and chemical parameters measured in waterbodies from HG.

| Parameters of waterbodies | Mean±SD | Range |
|---|---|---|
| Altitude | 495.07±114.2 | 337–828 |
| Water surface ($m^2$) | 42.8±79.5 | 2–400 |
| Depth (cm) | 30.8±24.9 | 15–150 |
| Temperature (°C) | 22.3±4.3 | 14–34 |
| pH | | 5.4–8.7 |
| Conductivity (µS/cm) | 220.6±149.6 | 34.2–683 |

**Table 5.** Model selection results. Models are ranked in a decreasing Akaike weight ($w_i$). Models with Akaike weight <0.05 are not shown. *K* - number of estimated parameters, AICc - second order Akaike Information Criterion corrected for small sample sizes, Δ*i* - AIC difference, $w_i$ - Akaike weight.

| | Model | *K* | *AICc* | Δ*i* | $w_i$ |
|---|---|---|---|---|---|
| Species richness | Mod4 | 3 | -20.19 | 0 | 0.65 |
| | Mod2 | 4 | -18.08 | 2.11 | 0.22 |
| | Mod8 | 3 | -14.88 | 5.31 | 0.05 |
| *B. bufo* | Mod2 | 3 | 50.27 | 0 | 0.21 |
| | Mod4 | 2 | 50.7 | 0.43 | 0.17 |
| | Mod7 | 3 | 50.87 | 0.6 | 0.15 |
| | Mod3 | 2 | 50.89 | 0.62 | 0.15 |
| | Mod8 | 2 | 52.36 | 2.09 | 0.07 |
| | Mod12 | 2 | 52.94 | 2.67 | 0.05 |
| | Mod11 | 2 | 53.11 | 2.84 | 0.05 |
| | Mod10 | 2 | 53.22 | 2.95 | 0.05 |
| | Mod9 | 2 | 53.25 | 2.98 | 0.05 |
| *B. variegata* | Mod4 | 2 | 46.22 | 0 | 0.29 |
| | Mod8 | 2 | 47.4 | 1.18 | 0.16 |
| | Mod2 | 3 | 47.8 | 1.58 | 0.13 |
| | Mod12 | 2 | 48.12 | 1.9 | 0.11 |
| | Mod11 | 2 | 48.73 | 2.51 | 0.08 |
| | Mod10 | 2 | 49.17 | 2.95 | 0.07 |
| | Mod3 | 2 | 49.31 | 3.09 | 0.06 |
| | Mod9 | 2 | 49.53 | 3.31 | 0.06 |
| *R. dalmatina* | Mod4 | 2 | 63.7 | 0 | 0.46 |
| | Mod2 | 3 | 65.66 | 1.96 | 0.17 |
| | Mod9 | 2 | 66.39 | 2.69 | 0.12 |
| | Mod11 | 2 | 67.47 | 3.77 | 0.07 |
| *R. temporaria* | Mod8 | 2 | 29.82 | 0 | 0.84 |
| *P. esculentus* complex | Mod11 | 2 | 62.99 | 0 | 0.56 |
| | Mod12 | 2 | 66.98 | 4 | 0.08 |
| | Mod3 | 2 | 67.42 | 4.43 | 0.06 |
| | Mod8 | 2 | 67.79 | 4.8 | 0.05 |
| | Mod9 | 2 | 67.98 | 5 | 0.05 |
| | Mod7 | 3 | 68.01 | 5.03 | 0.05 |
| *H. arborea* | Mod12 | 2 | 44.04 | 0 | 0.25 |
| | Mod4 | 2 | 45.22 | 1.18 | 0.14 |
| | Mod10 | 2 | 45.84 | 1.79 | 0.1 |
| | Mod8 | 2 | 45.9 | 1.86 | 0.1 |
| | Mod3 | 2 | 46 | 1.96 | 0.1 |
| | Mod9 | 2 | 46.12 | 2.08 | 0.09 |
| | Mod11 | 2 | 46.16 | 2.11 | 0.09 |
| | Mod7 | 3 | 46.84 | 2.8 | 0.06 |
| | Mod2 | 3 | 47.3 | 3.26 | 0.05 |
| *T. cristatus* | Mod4 | 2 | 45.19 | 0 | 0.34 |
| | Mod8 | 2 | 45.85 | 0.66 | 0.24 |
| | Mod12 | 2 | 46.76 | 1.57 | 0.15 |
| | Mod2 | 3 | 47.42 | 2.23 | 0.11 |
| *L. vulgaris* | Mod4 | 2 | 43.02 | 0 | 0.52 |
| | Mod2 | 3 | 44.15 | 1.12 | 0.29 |
| | Mod1 | 9 | 46.16 | 3.13 | 0.11 |

**Table 6.** Coefficients of covariates predicting species richness and individual species occurrence obtained by model averaging (significant associations are in *italics*). SE - unconditional standard error; CIL - unconditional lower confidence interval; CIU - unconditional upper confidence interval.

| | | Wch | Wv | Cc | Hiat | Wl | Iw | Alveg | Rl |
|---|---|---|---|---|---|---|---|---|---|
| Species richness | Estimate | 0.01 | *0.08* | 0.03 | 0.02 | -0.04 | 0.02 | -0.02 | 0.01 |
| | SE | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 |
| | CIL | -0.04 | 0.02 | -0.03 | -0.04 | -0.10 | -0.04 | -0.08 | -0.04 |
| | CIU | 0.06 | 0.13 | 0.08 | 0.07 | 0.01 | 0.07 | 0.03 | 0.07 |
| *B. bufo* | Estimate | -0.63 | 0.55 | -1.96 | 0.05 | 0.36 | 0.14 | 0.13 | 0.39 |
| | SE | 0.42 | 0.33 | 1.31 | 0.38 | 0.39 | 0.35 | 0.36 | 0.45 |
| | CIL | -1.45 | -0.10 | -4.52 | -0.71 | -0.41 | -0.55 | -0.58 | -0.50 |
| | CIU | 0.19 | 1.20 | 0.61 | 0.80 | 1.14 | 0.83 | 0.85 | 1.29 |
| *B. variegata* | Estimate | 0.32 | -0.65 | 0.14 | 0.23 | -0.63 | 0.50 | -0.30 | -0.45 |
| | SE | 0.41 | 0.34 | 0.47 | 0.40 | 0.41 | 0.53 | 0.35 | 0.33 |
| | CIL | -0.49 | -1.32 | -0.79 | -0.55 | -1.44 | -0.54 | -0.98 | -1.09 |
| | CIU | 1.13 | 0.03 | 1.06 | 1.00 | 0.18 | 1.53 | 0.38 | 0.19 |
| *R. dalmatina* | Estimate | -0.16 | *0.68* | 0.22 | -0.47 | -0.03 | -0.38 | -0.08 | -0.03 |
| | SE | 0.32 | 0.33 | 0.29 | 0.32 | 0.31 | 0.37 | 0.32 | 0.32 |
| | CIL | -0.79 | 0.03 | -0.34 | -1.09 | -0.63 | -1.11 | -0.71 | -0.65 |
| | CIU | 0.46 | 1.33 | 0.78 | 0.15 | 0.57 | 0.35 | 0.54 | 0.59 |
| *R. temporaria* | Estimate | 0.41 | 0.35 | -0.57 | 0.60 | *-1.48* | -0.47 | 0.24 | -0.81 |
| | SE | 0.53 | 0.47 | 1.04 | 0.75 | 0.64 | 0.92 | 0.68 | 1.34 |
| | CIL | -0.63 | -0.57 | -2.61 | -0.87 | -2.74 | -2.27 | -1.10 | -3.43 |
| | CIU | 1.45 | 1.26 | 1.46 | 2.07 | -0.23 | 1.34 | 1.57 | 1.80 |
| *P. esculentus* complex | Estimate | 0.36 | 0.16 | 0.53 | 0.29 | 0.37 | *0.74* | -0.03 | 0.40 |
| | SE | 0.33 | 0.30 | 0.32 | 0.34 | 0.36 | 0.33 | 0.34 | 0.31 |
| | CIL | -0.27 | -0.42 | -0.09 | -0.38 | -0.34 | 0.08 | -0.69 | -0.20 |
| | CIU | 1.00 | 0.74 | 1.15 | 0.95 | 1.09 | 1.39 | 0.63 | 1.00 |
| *H. arborea* | Estimate | 0.17 | 0.35 | 0.37 | 0.07 | -0.21 | -0.03 | -0.26 | -0.89 |
| | SE | 0.42 | 0.35 | 0.33 | 0.41 | 0.41 | 0.42 | 0.49 | 0.75 |
| | CIL | -0.65 | -0.33 | -0.27 | -0.74 | -1.01 | -0.84 | -1.22 | -2.37 |
| | CIU | 0.98 | 1.03 | 1.01 | 0.89 | 0.59 | 0.79 | 0.70 | 0.58 |
| *T. cristatus* | Estimate | -0.04 | *0.73* | 0.13 | -0.03 | -0.81 | 0.02 | 0.12 | 0.59 |
| | SE | 0.41 | 0.36 | 0.42 | 0.44 | 0.43 | 0.39 | 0.39 | 0.33 |
| | CIL | -0.85 | 0.04 | -0.69 | -0.90 | -1.66 | -0.74 | -0.64 | -0.07 |
| | CIU | 0.77 | 1.43 | 0.95 | 0.84 | 0.04 | 0.78 | 0.88 | 1.24 |
| *L. vulgaris* | Estimate | 0.44 | *1.25* | 0.09 | 2.86 | -1.74 | -0.04 | 1.23 | -1.04 |
| | SE | 0.49 | 0.59 | 0.49 | 2.03 | 1.16 | 0.76 | 0.96 | 1.73 |
| | CIL | -0.52 | 0.10 | -0.87 | -1.11 | -4.00 | -1.52 | -0.65 | -4.43 |
| | CIU | 1.40 | 2.41 | 1.06 | 6.83 | 0.53 | 1.44 | 3.11 | 2.36 |

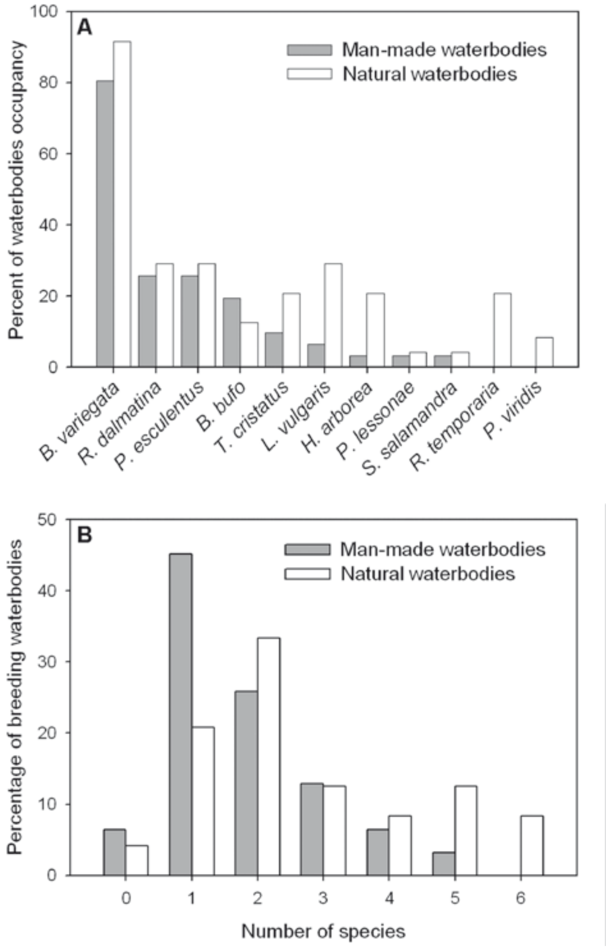


**Fig. 2** (A) Frequency of amphibian species in waterbodies with natural (*n*=24) and man-made origin (*n*=31). (B) Amphibian species richness in waterbodies of natural and man-made origin (*n*=55).

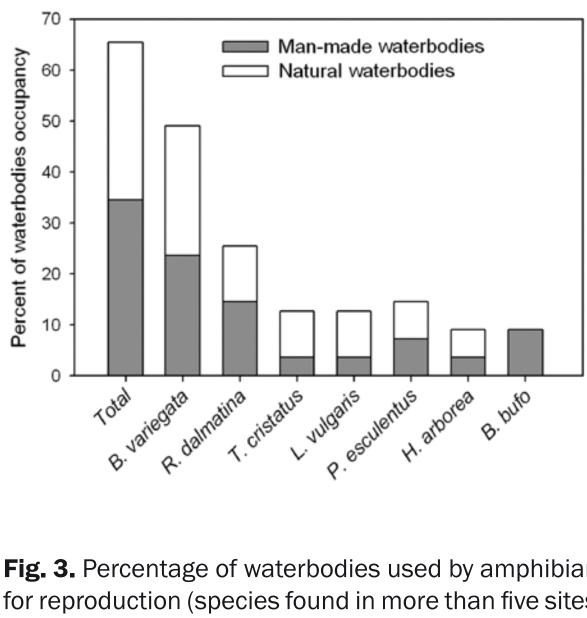


**Fig. 3.** Percentage of waterbodies used by amphibians for reproduction (species found in more than five sites).

# DISCUSSION

A high percentage of the inventoried waterbodies used by amphibians were man-made: temporary ponds such as puddles and ponds near the road, drainage ditches including open ditches parallel to the road or traditional agricultural drainage ditches. Almost 75% of such waterbodies possessed a high risk of desiccation, and were favoured by species with a short larval period, such as *B. variegata*, *R. dalmatina* and *P. esculentus* complex. *Bombina variegata* was the most common species in the region, despite its high protection status.

Man-made waterbodies play an important role in the persistence of amphibians (Knutson et al., 2004; Dalbeck & Weinberg, 2009), sometimes providing the only available breeding habitats (e.g. Curado et al., 2011). Our research confirms these findings. Only two species were not detected in man-made habitats (*R. temporaria* and *P. viridis*). The low occurrence of *R. temporaria* might be caused by a study site at low elevation. More than a third of the waterbodies contained reproductive adults, but no eggs or larvae. Adult individuals may have used such water bodies as stepping stones (e.g. Hartel et al., 2011), connecting critical habitats for breeding (Semlitsch, 1998; Mazerolle, 2005).

Aquatic habitat characteristics were generally more important than landscape parameters. This finding is in line with other studies on amphibian habitat use traditional rural landscapes of Romania (Hartel et al., 2010a, b). The quality of waterbodies may change rather rapidly, while the terrestrial habitat quality remains more constant due to the persistence of traditional agriculture. Precipitation is a major factor influencing the quality of temporary ponds, and hydroperiod can be the most important determinant of waterbody use where only temporary ponds are available for breeding (e.g. Hartel et al., 2011). Permanent ponds often contain introduced fish predators which are harmful for some amphibian species (Hartel et al., 2010c).

Models containing terrestrial habitat variables were generally stronger predictors of amphibian occurrence in ponds. This highlights the importance of considering habitat complementation (i.e. both aquatic and terrestrial habitats) for understanding occurrence of pond breeding amphibians (van Buskirk, 2005). *Pelophylax esculentus* complex appeared frequently in ponds near human settlements. This species may colonize very heterogeneous habitats (Pavignano et al., 1990; Pagano et al., 2001) and can be found in landscapes highly exploited by humans (Ficetola & De Bernardi, 2004; Loman & Lardner, 2006).

## Implications for conservation

Nine out of eleven amphibian species used man-made waterbodies either for reproduction or dispersal. After 1990, changes in land ownership produced changes in land use patterns including land abandonment and land use intensification (Boboiciov & Cojocaru, 2010). Both can result in the loss or isolation of waterbodies. In our study area, 73% of the inventoried waterbodies had agricultural fields within 500 m of their surroundings. *Bombina variegata* declined in many parts of Western Europe following the disappearance of small, temporary waterbodies (Stumpel, 2004), making it rather sensitive to landscape alteration. Man-made habitats including ponds may depend on continuous maintenance activities (Curado et al., 2011; Garcia-Gonzalez & Garcia-Vazquez, 2011), and they will disappear when no longer needed or when the management activities that created them stop. Actually, traditional activities realised in HG contribute to the maintaining of amphibian breeding habitats. Cattle and sheep grazing maintain temporary ponds because control vegetation development (which can in turn shorten the hydroperiod of the temporary ponds).. Carts create and maintain temporary ponds along dirt roads, which are important habitats for *B. variegata*. The promotion of environment-friendly agricultural practices is a key factor for the maintenance of amphibians in the studied landscapes.

## ACKNOWLEDGEMENTS

The present work is a part of a research project funded by a Rufford Foundation Small Grant to DC. The Administration of the Haţeg Geopark provided access permits and logistic support in the field. We are grateful to three anonymous reviewers for their constructive comments on the manuscript. The work of Tibor Hartel was supported by the strategic grant POSDRU/89/1.5/S/58852, Project "Postdoctoral program for training scientific researchers" co-financed by the European Social Found within the Sectorial Operational Program Human Resources Development 2007–2013.

## REFERENCES

Abellan, P., Sanchez-Fernandez, D., Millan, A., Botellan, F., Sanchez-Zapata, J.A. & Gimenez, A. (2006). Irrigation pools as macroinvertebrate habitat in a semi-arid agricultural landscape (SE Spain). *Journal of Arid Environments* 67, 255–269.

Baillie, J.E.M., Bennun, L.A., Brooks, T.M., Butchart, S.H.M., Chanson, J.S., Cokeliss, Z., Hilton-Taylor, C., Hoffmann, M., Mace, G.M., Mainka, S.A., Pollock, C.M., Rodrigues, A.S.L., Stattersfield, A.J. & Stuart, S.N. (2004). *2004 IUCN Red List of Threatened Species. A Global Species Assessment*. Gland, Switzerland and Cambridge, UK: IUCN.

Beebee, T.J.C. (1997). Changes in dewpond numbers and amphibian diversity over 20 years on chalk downland in Sussex, England. *Biological Conservation* 81, 215–219.

Bignal, E.M. & McCracken, D.I. (2000). The nature conservation value of European traditional farming systems. *Environmental Reviews* 8, 149–171.

Boboiciov, S. & Cojocaru, L. (2010). Suppositions in order of reconsideration of the abandoned land – Meadows and former agricultural land. *Research Journal of Agricultural Science* 42 (1), 363–366.

Burnham, K.P. & Anderson, D.R. (2002). *Model selection and multimodel inference: a practical information-theoretic approach*. 2nd Edition. Springer-Verlag, USA.

Ciaian, P. & Pokrivcak, J. (2007). Agriculture Reforms and Development in East-Central Europe, 117–132. In *Industries and Markets in Central and Eastern Europe*. Sergi, B.S., Bagatelas, W.T. & Kubicova, J. (eds). Aldershot; Burlington: Ashgate.

Ćirović, R., Vukov, T.D., Radović, D., Džukić, G. & Kalezić, M.L. (2008). Distribution patterns and environmental determinants of European newts in the Montenegrin karst area. *Biologia* 63 (5), 745–752.

Curado, N., Hartel, T. & Arntzen, J.W. (2011). Amphibian pond loss as a function of landscape change - a case study over three decades in an agricultural area of northern France. *Biological Conservation* 5, 1610–1618.

Dalbeck, L. & Weinberg, K. (2009). Artificial ponds: a substitute for natural Beaver ponds in a Central European Highland (Eifel, Germany)? *Hydrobiologia* 630 (1), 49–62.

Diniz-Filho, J.A.F., Sant'Ana, C.E.R. & Bini, L.M. (1998). An eigenvector method for estimating phylogenetic inertia. *Evolution* 52, 1247–1262.

ESRI (1995–2010). *ArcGIS Desktop*. Environmental Systems Research Institute, Redlands CA, USA.

European Environment Agency (2009). *Corine land cover 2000 (CLC2000) seamless vector database*. http://www.eea.europa.eu/data-and-maps/data/corine-land-cover-2000-clc2000-seamless-vector-database-1 (accessed on September 20, 2009).

Eurostat (2009). *Europe in figures – Eurostat yearbook 2009*. http://epp.eurostat.ec.europa.eu/portal/page/portal/publications/eurostat_yearbook (accessed on December 1, 2009).

Ficetola, G.F. & De Bernardi, F. (2004). Amphibians in a human-dominated landscape: the community structure is related to habitat features and isolation. *Biological Conservation* 119, 219–230.

Garcia-Gonzalez, C. & Garcia-Vazquez, E. (2011). The value of traditional troughs as freshwater shelters for amphibian diversity. *Aquatic Conservation: Marine and Freshwater Ecosystems* 21, 74–81.

Hartel, T., Öllerer, K., Cogălniceanu, D., Nemes, Sz., Moga, C.I. & Demeter, L. (2010a). Using a habitat based approach in mapping amphibian distribution in Romania: study case of the Saxon cultural landscape in the Târnava Mare basin. *Italian Journal of Zoology* 77, 61–70.

Hartel, T., Schweiger, O., Öllerer, K., Cogalniceanu, D. & Arntzen, J.W. (2010b). Amphibian distribution in a traditionally managed rural landscape of Eastern Europe: probing the effect of landscape composition. *Biological Conservation* 143, 1118–1124.

Hartel, T., Nemes, Sz., Öllerer, K., Cogalniceanu, D., Moga, C. I., Arntzen, J.W. (2010c). Using connectivity metrics and niche modeling to explore the occurrence of the Northern crested newt (Amphibia, Caudata) in a traditionally managed landscape. *Environmental Conservation* 37, 195–200.

Hartel, T., Băncilă, R., Cogălniceanu, D. (2011) Spatial and temporal variability of aquatic habitat use by amphibians in a hydrologically modified landscape. *Freshwater Biology* 56, 2288–2298.

Hazell, D., Hero, J.M., Lindenmayer, D. & Cunningham, R. (2004). A comparison of constructed and natural habitat for frog conservation in an Australian agricultural landscape. *Biological Conservation* 119, 61–71.

Hels, T. & Buchwald, E. (2001). The effect of road kills on amphibian populations. *Biological Conservation* 99, 331–340.

Herzon, I. & Helenius, J. (2008). Agricultural drainage ditches, their biological importance and functioning. *Biological Conservation* 141, 1171–1183.

Heymann, Y., Steenmans, C., Croissille, G. & Bossard, M. (1994). *CORINE Land Cover technical guide*. Luxembourg: Office for Official Publications of the European Communities.

Jackson, D.A. (1993). Stopping rules in principal component analysis: a comparison of heuristical and statistical approaches. *Ecology* 74, 2204–2214.

Knutson, M.G., Richardson, W.B., Reineke, D.M., Gray, B.R., Parmelee, J.R. & Weick, S.E. (2004). Agricultural ponds support amphibian populations. *Ecological Applications* 14, 669–684.

Kuemmerle, T., Müller, D., Griffiths, P. & Rusu, M. (2009). Land use change in Southern Romania after the collapse of socialism. *Regional Environmental Change* 9, 1–12.

Loman, J. & Lardner, B. (2006). Does pond quality limit frogs *Rana arvalis* and *R. temporaria* in agricultural landscapes? A field experiment. *Journal of Applied Ecology* 43, 690–700.

Mazerolle, M.J. (2005). Drainage ditches facilitate frog movements in a hostile landscape. *Landscape Ecology* 20, 579–590.

Mazerolle, M.J. & Desrochers, A. (2005). Landscape resistance to frog movements. *Canadian Journal of Zoology* 83, 455–464.

Mazerolle, M.J. (2009). *AICcmodavg: Model selection and multimodel inference based on (Q)AIC(c)*. http://cran.r-project.org/web/packages/AICcmodavg/index.html (accessed on July 4, 2009).

Moran, N.A. (1994). Adaptation and constraint in the complex life cycles of animals. *Annual Review of Ecology and Systematics* 25, 573–600.

Pagano, A. & Joly, P. (1998). Limits of the morphometric method for field identification of water frogs. *Alytes* 16, 15–23.

Pagano, A., Crochet, P.A., Graf, J.D., Joly, P. & Lodé, T. (2001). Distribution and habitat use of water frog hybrid complexes in France. *Global Ecology and Biogeography* 10, 433–441.

Palang, H., Printsmann, A., Gyuró, E.K., Urbanc, M., Skowronek, E. & Woloszyn, W. (2006). The forgotten landscapes of Central and Eastern Europe. *Landscape Ecology* 21, 347–357.

Pavignano, I., Giacoma, C. & Castellano, S. (1990). A multivariate analysis of amphibian habitat determinants in north western Italy. *Amphibia-Reptilia* 11, 311–324.

Plieninger, T., Höchtl, F. & Spek, T. (2006). Traditional land-use and nature conservation in European rural landscapes. *Environmental Science & Policy* 9, 317–321.

R Development Core Team (2009). *R: A language and environment for statistical computing. R Foundation for Statistical Computing*. Vienna. (http://www.R-project.org; accessed on July 3, 2009).

Ruggiero, A., Cereghino, R., Figuerola, J., Marty, P. & Angelibert, S. (2008). Farm ponds make a contribution to the biodiversity of aquatic insects in a French agricultural landscape. *Comptes Rendus Biologies* 331, 298–308.

Semlitsch, R.D. (1998). Biological delineation of terrestrial buffer zones for pond-breeding salamanders. *Conservation Biology* 12, 1113–1119.

Stuart, S.N., Chanson, J.S., Cox, N.A.,Young, B.E., Rodrigues, A.S.L., Fischman, D.L. & Waller, R.W. (2004). Status and trends of amphibian declines and extinctions worldwide. *Science* 306, 1783–1786.

Stumpel, A.H.P. (2004). *Reptiles and amphibians as targets for nature management*. Alterra Scientific Contributions No.13, Alterra, Wageningen.

Thiere, G., Milenkovski, S., Lindgren, P.E., Sahlén, G., Berglund, O. & Weisner, S.E.B. (2009). Wetland creation in agricultural landscapes: bio-diversity benefits on local and regional scales. *Biological Conservation* 142, 964–973.

Turnok, D. (1996). The Rural Transition in Romania. *Eastern European Countryside* 2, 45–57.

Van Buskirk, J. (2005). Local and landscape influence on amphibian occurrence and abundance. *Ecology* 86, 1936–1947.

Wells, K.D. (2007). *The ecology and behavior of amphibians*. Chicago, IL: University of Chicago Press.

Young, J., Richards, C., Fischer, A., Halada, L., Kull, T., Kuzniar, A., Tartes, U., Uzunov, U. & Watt, A. (2007). Conflicts between biodiversity conservation and human activities in the Central and Eastern European Countries. *Ambio* 36, 545–550.

*Accepted: 17 July 2011*