

Maintenance of biodiversity in vineyard-dominated landscapes: a case study on larval salamanders

Matteo Tanadini¹, Benedikt R. Schmidt^{2,3}, Pierre Meier⁴, Jérôme Pellet^{2,5} & Nicolas Perrin¹

¹ Department of Ecology and Evolution, University of Lausanne, Lausanne, Switzerland

² KARCH, Neuchâtel, Switzerland

³ Institute of Evolutionary Biology and Environmental Studies, University of Zurich, Zürich, Switzerland

⁴ Service de la consommation et des affaires vétérinaires, Lausanne, Switzerland

⁵ A. Maibach Sàrl, Oron-la-Ville, Switzerland

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Correspondence

Matteo Tanadini, Viktoriastrasse 55, 8050 Zürich, Switzerland. Tel: +41 91 791 60 31; Fax: +41 91 791 60 31
Email: matteo.tanadini@gmail.com

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Abstract

Alteration of natural habitats as a result of agricultural intensification is detrimental for wildlife. There is, however, growing evidence that land use and management can be wildlife friendly. In Europe, agricultural areas cover two-thirds of the land and therefore play a major role in maintaining biodiversity. Agricultural land use is very intense in vineyard-dominated landscapes but there are no refuges for wildlife in the form of ecological compensation areas. In our study, we assessed spatial variation in abundance of salamander (*Salamandra salamandra*) larvae in relation to land use and stream characteristics in vineyard-dominated landscapes. Abundance of larval salamanders depended positively on weirs, amount of riparian vegetation along the streams and environment-friendly agricultural practice in the vineyards. Surprisingly, road density also had positive effects, presumably through indirect effects (stone walls along roads may serve as refugia). Thus, abundance is determined by characteristics of both the aquatic and terrestrial habitats. Our results suggest that fire salamanders can persist in landscapes dominated by intensive agriculture like viticulture, indicate wildlife-friendly management options and highlight that man-made habitat can be valuable for wildlife.

Introduction

European landscapes have been shaped by human since millennia (Pullin *et al.*, 2009), this created a great diversity of so-called cultural landscapes with a high biodiversity (Pimentel *et al.*, 1992). Agricultural landscapes cover two-thirds of the European land and therefore play an important role for biodiversity conservation. Unfortunately, the intensification of agriculture in Europe over the past decades led to a decline in farmland biodiversity (Pimentel *et al.*, 1992; Donald, Green & Heath, 2001; Kleijn *et al.*, 2009) which also affected amphibians (Beja & Alcazar, 2003). This negative trend may be reversible because several studies showed that simple changes in farming practice sometimes enhance biodiversity in agricultural land (Peach *et al.*, 2001; Van Buskirk & Willi, 2004; Kleijn *et al.*, 2006; Merckx *et al.*, 2009). These studies highlight the fact that a species-rich fauna and flora can live in modern man-made agricultural habitats and landscapes (Knutson *et al.*, 2004; Aviron *et al.*, 2009). It is therefore important to know which landscape features allow species to persist in agricultural landscapes.

We studied stream and landscape features that may allow populations of the stream-breeding fire salamander (*Salamandra salamandra*), a species usually associated with

natural landscapes (Ficetola *et al.*, 2011), to persist in vineyard-dominated agricultural landscapes. We selected vineyards because environment-friendly farming practices like organic farming have little effect on biodiversity in vineyards (Bruggisser, Schmidt-Entling & Bacher, 2010), and because there are no set-asides in vineyards that may serve as habitat for wildlife (Van Buskirk & Willi, 2004). We aimed to produce evidence-based recommendations for farmers (winemakers) to protect salamanders while maintaining agricultural productivity.

Methods

Study area

We conducted surveys in the two largest vineyard-dominated regions in Switzerland: Lavaux and La Côte on the northern shore of the Lake Geneva. These areas are a matrix of vineyards (about 2/3) and urban land (about 1/3).

Stream partitioning

We surveyed all 17 fishless streams present in the two regions. During a diurnal survey, streams were divided into

homogeneous sections using a standard protocol (BUWAL, 1998). Based on four morphological features (bed width variation, naturalness of the stream bed, naturalness of stream banks and naturalness of riparian vegetation), 65 sections among 17 streams were retained. All streams were first Strahler stream order, which fire salamanders seem to prefer (Thiesmeier & Günther, 1996). The minimal distance between any two streams was 90 m. Mean distance between adjacent streams is $1.2 \text{ km} \pm 1.3$.

Data collection

Female *S. salamandra terrestris* gives birth to larvae in spring (Thiesmeier & Günther, 1996). Streams were visited from 22 April to 26 June 2009 after the peak deposition of larvae. We performed one-night survey for each stream; larvae being more active at night. Streams were visited upwards in a randomly chosen order. We counted all visible larvae and calculated the apparent larval abundance as the number of larvae per 10 m of stream. In the experimental assessment of Jung *et al.* (2002), counts of larval amphibians in simple aquatic environments were highly correlated with true abundance ($R^2 > 0.66$; Schmidt, 2004). Stream features that could affect detectability of salamander larvae were similar among all streams. Pools were less than 50 cm deep enabling a good visibility of the stream bottom. There was little aquatic vegetation or algae. The stream bed was mainly composed of gravel and rocks, while woody debris and leaves were only infrequently encountered. To test whether counts were repeatable, we conducted a second survey at six streams. The result was that counts were similar. To account for spatiotemporal variation in larvae abundance (through mortality or drift), we included the variable date as a random effect.

Predictor variables

Because the distribution of salamanders can depend on both the aquatic and the terrestrial habitats (Ficetola *et al.*, 2011), we analysed larval salamander densities using predictor variables measured at three spatial scales: at the section level, at the stream level and at the metapopulation level. There were two section level variables: the frequency of artificial weirs along sections which modify structural and hydrological properties of streams (hereafter 'weirs') and the amount of woody riparian vegetation along the stream (expressed as a proportion of the length of the section; 'vegetation'). Weirs and vegetation were measured in the field. We measured four variables at the stream level. We either used a 400 m buffer along the stream or quantified predictor variables for the entire watershed; the choice of 400 m is based on the results presented in Ficetola, Padoa-Schioppa & De Bernardi (2009). Grass removal, a frequent practice in viticulture, determines the photosynthetic biomass present within vineyards. Therefore, we calculated the mean normalized difference vegetation index ('NDVI'; Gates, 1980) for all parcels within the buffer defined as vineyards. To do so, we used the Swiss land use

vector database Vector 25 (Swisstopo, 2000) and satellite images with a 30-m resolution (<http://glcf.umiacs.umd.edu> date of access: 20 October 2009; picture 036-259 taken on 21 July 2001, and picture 220-860 taken on 21 September 2006). In the study area, the practice of grass removal has not changed over the last decade. The percentage of impervious surface ('%IS'; defined as settlements and roads) which determines water runoff on land and the density of major roads ('roads', i.e. all two-lane roads) were quantified using the Swiss land use database Vector 25. Fungicide use was indirectly quantified using the mean copper content in the moss *Fontinalis antipyretica* ('copper'). Copper is one of the major components of fungicides employed in viticulture (BUWAL, 2003). This aquatic moss is widely used as bio-indicator for heavy metal loads in lotic ecosystems because of its absorption properties (Bleuel *et al.*, 2005). We calculated an average copper level for every stream on the basis of three moss samples collected during August 2009. Samples were analysed as described by Klein, Meier & Aubert (1991). NDVI and roads were measured within 400-m buffers whereas %IS and copper were measured within watersheds.

The metapopulation-level variable was connectivity ('connectivity'; Prugh, 2009), which was computed for each stream as follows:

$$\text{connectivity}_i = \sum_{j \neq i} e^{\alpha d_{ij}} Q_j \quad (1)$$

where Q_j is the mean larval abundance in stream j , d_{ij} is the distance between centroids of the focal stream i and stream j in km. Finally, $1/\alpha$ is the species mean dispersal distance. Based on Ficetola, Padoa-Schioppa & De Bernardi (2009), we set this parameter to 0.4 km.

Three variables were included to control for temporal and spatial variation. We included date of survey in the model ('date'), because larval densities decrease after the initial peak of deposition due to mortality and drift. The second control variable is the position of the focal section in the stream ('position'), because conditions of a given section depend on the conditions of the sections placed above that focal section. Finally, stream slope ('slope') was added as control because it defines hydraulic conditions, which has been shown to be a major determinant of larval abundances (Baumgartner, Waringer & Waringer, 1999). As further control variables, we included in the model the random effect region ('region'). As sections belonging to the same stream are not independent, we also included the random effect stream ('stream').

All predictors were measured in the field or used the geographical information system ArcMap GIS 9.3 (ESRI, Redlands, CA, USA). One person (Matteo Tanadini) performed the ecomorphological assessments (division of streams in sections), evaluated larval abundances and collected bryophyte samples.

All possible two-way interactions between predictors were inspected graphically. As a result, we added the interactions vegetation:%IS, vegetation:NDVI, copper:NDVI, NDVI:%IS, roads:NDVI and copper:roads to the model.

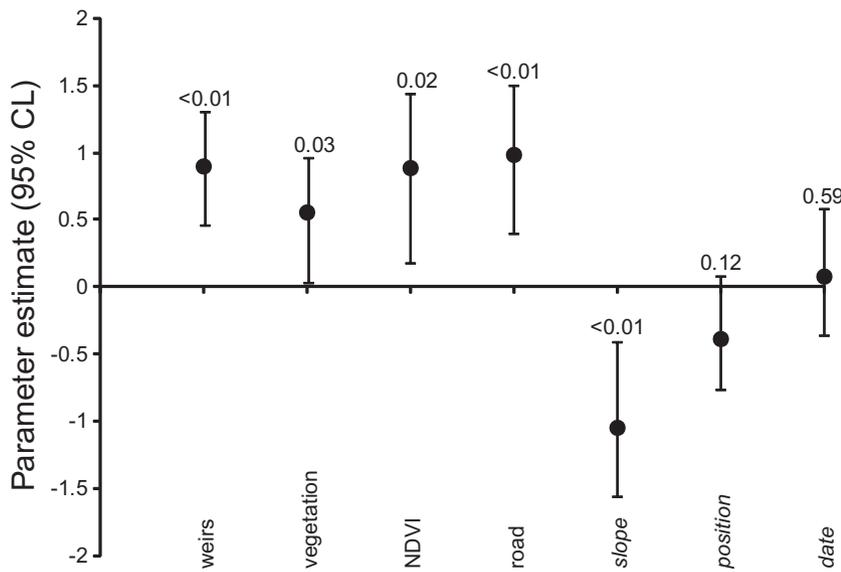


Figure 1 Parameter estimates, confidence limits (CLs) and *P*-values of the parameters retained in the final model. Random effects are not shown. Variables that serve as controls for spatial and sampling variation are printed in italics. NDVI, normalized difference vegetation index.

Statistical analyses

We analyzed fire salamander abundances using generalized linear mixed models (Bolker *et al.*, 2009). The Pearson correlation coefficient between the 10 predictor variables never exceeded 0.6, therefore we considered the risk of multicollinearity to be limited (Berry & Feldman, 1990).

Exploratory graphs identified an extreme value that was removed from the dataset (larval abundance = 128). After removing this data point, 64 sections belonging to 17 streams were retained for statistical analysis.

The response variable (larval abundance) was square root transformed to improve normality of residuals. All predictor variables were standardized.

We built a model with the following formula: $\text{lmer}[\sqrt{\text{larval_density}} \sim \text{predictors (fixed effects)} + \text{control variables (fixed effects)} + \text{interactions retained (fixed effects)} + (1|\text{stream}) + (1|\text{region})]$ (random effects). We then used a backward stepwise approach to reduce the number of predictor variables. Using Markov Chain Monte Carlo (50 000 simulations), a *P*-value was calculated for each variable contained in the model. Control variables were kept in the model until the end. All statistical analyses were performed with R 2.10.0 using the packages lme4 and LanguageR (R Development Core Team, 2009).

Results

Salamander larvae were present in all 64 stream sections. Median density was 26 larvae per 10 m of stream (range: 0.6–70). The final model explained 59.8% of the total variance and included four predictors and the control variables, namely the density of weirs, the amount of riparian vegetation, the mean plant biomass in vineyards and the density of major roads within a 400-m buffer. All had a positive effect on larval salamander abundance (Fig. 1).

Discussion

The four predictors retained in the model describe both the stream and surrounding terrestrial habitat (similar to the results of Ficetola *et al.*, 2011). The result indicates that landscape complementation (Pope, Fahrig & Merriam, 2000) is important for the fire salamander and suggests ways to conserve stream-associated wildlife in vineyards, a type of agricultural land use for which there are no set-asides and for which organic farming does not appear to be beneficial for wildlife (Bruggisser, Schmidt-Entling & Bacher, 2010).

Even though human alteration of streams can negatively affect amphibian communities (Hazell, Osborne & Lindenmayer, 2003), the positive effect of weirs can be easily interpreted. These man-made structures create pools and other areas of low hydraulic stress. Pools within artificialized streams are known to harbor high densities of salamander larvae (Baumgartner *et al.*, 1999), probably because they reduce larval drift or act as refuges for drifting larvae. Thus, human alteration of natural environment can result in the creation of good secondary habitats for wildlife (Hazell *et al.*, 2004; Knutson *et al.*, 2004; Denoël & Lehmann, 2006). In the present case, the positive effect of stream alteration may be caused by the fact that adding weirs created chains of pools which are known to be good habitat for amphibians (Hazell *et al.*, 2003).

Riparian vegetation had beneficial effects confirming previous studies on stream-breeding salamanders (Crawford & Semlitsch, 2007). Even a narrow strip of riparian vegetation (in our case 5–20 m wide) may have beneficial effects by providing terrestrial habitat, hibernacula or corridors for movements. It may provide coarse woody debris in the stream and buffer streams against the impact of surrounding human activities (Allan, 2004; Crawford & Semlitsch, 2007).

The positive effect of mean plant biomass in vineyards might be direct (by providing food or shelter) or indirect.

Low NDVI may indicate an intensive practice associated with heavy use of herbicides with negative effects on salamander larvae (Rohr *et al.*, 2006). Alternatively, overland water flow is known to be higher on bare soils compared with vegetated soils (Maidment, 1993) which may lead to stronger spates (Paul & Meyer, 2001) or affect phosphate concentrations in the stream (Ficetola *et al.*, 2011).

Unexpectedly, road density had a positive effect, even though roads are usually detrimental for amphibians (Van Gelder, 1973; Hels & Buchwald, 2001). Such effects might be indirect, however (Zanini *et al.*, 2008), if road density correlates with unmeasured positive factors. Stone walls that are often built along roads and known to serve as refuges for adult salamanders (Rebello & Leclair, 2003) might be such a factor.

We offer four main conclusions. First, larval densities were best explained by a model that included both stream and landscape variables. This confirms that landscape complementation is important for amphibians (Pope *et al.*, 2000; Semlitsch & Bodie, 2003; Ficetola *et al.*, 2011) and supports their conclusion that amphibian conservation can only be successful if both aquatic and terrestrial habitats are preserved (Cushman, 2006). Second, salamanders can persist even in a high-intensity agricultural landscape without an agri-environment scheme (i.e. set-asides). Salamander larvae were found in all streams and abundances were higher than those observed in natural ecosystems (Baumgartner *et al.*, 1999). Third, there are simple means by which farmers can make their property more salamander friendly. Allowing trees and bushes to grow along streams benefits salamanders and probably wildlife in general. This would result in vegetated buffer zones along streams. The creation of such zones is now the focus of the new regulations on water courses protection by the Swiss federal government. Salamanders, and most likely other wildlife as well, would also benefit if farmers allow grass cover within vineyard parcels. Fourth, some habitat modifications (weirs) positively affected salamanders even though the species is described as a species that is more commonly found in natural landscapes (Ficetola *et al.*, 2011). This highlights the fact that man-made habitat can replace natural habitat, at least partially. Of course, where there still is pristine natural habitat, pristine habitat should be preserved. However, where pristine habitat is destroyed and replaced by man-made habitat, then this new habitat can be valuable for threatened species and should not be ignored by conservationists viewing only pristine natural habitat as valuable. Some species may occur nowadays only in man-made habitat [e.g. natterjack toads in gravel pits (Meisterhans & Heusser, 1970)]. If a species has lost its natural habitat and conservationists ignore that the species can survive in man-made habitat (Stevens & Baguette, 2008), then the species would be doomed to extinction.

Finally, we end with a cautionary note because correlation is not causation. We found that roads had a positive effect on salamanders. Yet, adult salamanders are often killed on roads and this should negatively affect populations (Schmidt, Feldmann & Schaub, 2005). This leads us to con-

clude that the results of statistical models fitted to non-experimental data should not be accepted uncritically (Anderson *et al.*, 2001). In order to better understand cause-effect relationships, a combination of experimental and observational studies would clearly allow stronger inference (Werner, 1998; Hooper *et al.*, 2008).

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