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<AT>On the importance of taking into account agricultural practices when defining conservation priorities for regional planning
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<ABS-HEAD>Highlights ► Systematic conservation planning is increasingly used to inform regional land planning ► Including overlapping stressors (e.g. urbanization) does not change conservation priorities ► Including non-overlapping stressors (e.g. agriculture) changes conservation priorities ► There is a need for a better understanding of the impact of agriculture on biodiversity

<ABS-HEAD>Abstract

<ABS-P>Conserving biodiversity in managed landscapes requires the definition of spatial conservation priorities. The systematic conservation planning tools which are used to define these conservation priorities, assess the vulnerability of different locations by combining two different elements: some measurement of the biological assets in question, and some measurement of the key processes which threaten these biological assets. For instance, in cumulative impact mapping, maps of individual human activities that impact ecosystems (hereafter referred to as ‘stressor’ for individual maps and ‘cumulative stressor’ for combined maps) are overlaid with maps of ecosystem vulnerability, in order to estimate the overall ecological impact of human activities on natural ecosystems. These tools are appealing because they are easy to use and inform regional land planning. However, given that once these spatial conservation priorities are defined they potentially have far-reaching consequences, there is a need to test their robustness and reliability. Here we propose to investigate how the uncertainties related to the estimation of a cumulative stressor layer affect the definition of spatial conservation priorities. We conduct a sensitivity analysis of the different ways of estimating major stressors related to human activities (transport, urbanization and population) with a specific focus on agriculture. We show that spatial conservation priorities are little sensitive to most of the parameters and input data used to estimate the cumulative stressor map. In particular, they are not very sensitive to changes in

spatially overlapping stressors, i.e. those which overlap spatially with other stressors. However, our analyses also reveal that spatial conservation priorities are highly sensitive to how the agriculture stressor is defined. These results highlight the importance of better understanding how agricultural activities impact biodiversity and establishing how more accurate information on agricultural practices can be used to define spatial conservation priorities.

<KWD>Keywords: Biodiversity conservation; Agricultural practices; Human footprint; GIS; Land planning; Green corridors

Introduction

Both land use and climate change, as major threats to terrestrial biodiversity, mean that environmental policies need to focus regional land planning on the conservation of natural habitats and their connectivity (Crooks & Sanjayan, 2006; Heller & Zavaleta, 2009, Seto et al., 2012). A wide variety of tools are now available to help define spatial conservation priorities (Wilson et al., 2009), i.e. the areas where investment in conservation actions will be most effective (Margules & Pressey, 2000). In systematic conservation planning, these tools are typically used to estimate the overall ecological impact of human activities on natural ecosystems and assess the vulnerability of different locations and/or biological assets by combining two different measurements for each location (Brooks, 2006; Redford et al., 2003; Theobald & Hobbs, 2002; Wilson et al., 2005; Wilson et al., 2009 Figure 1). The first set of measurements describe the biological assets or biodiversity facets we hope to conserve, generally using biodiversity surrogates (e.g. number of species, habitat types, Ferrier, 2002), to identify areas with high biodiversity value ('hotspots'). The second set of measurements

describe the potential intensity and impact of key threatening processes related to human activities (also called threats, or pressures, hereafter ‘stressor’ for individual maps) that affect those areas or biological assets. Cumulative stressor layers (Halpern & Fujita, 2013; Halpern et al., 2008) are obtained by summing individual stressor maps (Sanderson et al., 2002; Woolmer et al., 2008) and can thus be used to account for the high spatial variability in stressors (Wilson et al., 2005) and for multiple pressures simultaneously (Oliver et al., 2014). This principle has been applied to both marine (Coll et al., 2010; Halpern et al., 2008) and terrestrial ecosystems (Bellard et al., 2015; Evans et al., 2011).

In order to ensure policy and decision-making are not misled by the definition of spatial conservation priorities, there is an obvious need to address and clearly communicate the uncertainties and caveats relating to these existing tools (Halpern & Fujita, 2013). For instance, different sources of uncertainties relating to the estimation of the cumulative stressor layer could affect the definition of spatial conservation priorities. Indeed, the mapping of this layer remains somewhat arbitrary given that expert knowledge or broad literature reviews are used to define which individual stressors need to be taken into account and to assess the intensity and potential impact of each stressor on biodiversity assets (e.g. the impact of transport infrastructures on biodiversity is well documented: Forman 2003; Marcantonio et al., 2013). When dealing with terrestrial ecosystems, the role of agriculture as a potential threat to terrestrial biodiversity is particularly difficult to assess, for three main reasons. Firstly, agriculture is expected to have both positive and negative effects on biodiversity depending on the practices and spatial configuration used, and should consequently be considered as both a component of, and a threat to, biodiversity (Bassa et al., 2011; Batáry et al., 2012; Kleijn et al., 2006). Secondly, different facets of biodiversity (e.g. species abundance, species richness, functional diversity) and different types of organisms respond differently to agricultural practices (Gabriel et al., 2010; Ponce et al., 2011). Thirdly,

agriculture is often described using broad crop types as a proxy, while agricultural practices seem to be what really matter (e.g. the maintenance of spatial heterogeneity, the use of chemicals, organic practices; Arndorfer et al., 2012).

Running a sensitivity analysis on the tools used to define spatial conservation priorities can be useful for identifying the parameters and input data with the greatest impact on land planning outcomes (Wilson et al., 2005). Here we propose to investigate how the uncertainties relating to the estimation of a cumulative stressor layer affect the definition of spatial conservation priorities. We test the effects of both the general parameterization and of how the agriculture stressor layer is defined (i.e. the stressor layer related to the impact of agricultural practices on biodiversity). Within the framework proposed by Vimal et al. (2012, Figure 1), we analyze how different ways of mapping cumulative stressor layers affect the definition of conservation priorities in Mediterranean France. This framework was developed in order to produce the spatial regional priorities required by the French government for the implementation of the green and blue national corridors (SRCE PACA, 2013). Layers describing biodiversity assets and cumulative stressors are directly compared and contrasted (Figure 1). Highest conservation priority is given to areas with both high levels of biodiversity and of cumulative stressors; low priority is given to areas with low levels of biodiversity (or a medium level of biodiversity but low cumulative stressors); and intermediate priority is given to the remaining cases (Figure 1). Our analyses target four main stressors known to affect terrestrial biodiversity: transport infrastructure, urbanization, agriculture and human population density.

Methods

The study region

This study was carried out in Mediterranean France, which covers two administrative regions (Provence-Alpes-Côte d'Azur and Languedoc-Roussillon) in Southern France (58,700 km², see Supplementary Material Figure S1). Benefitting from strong environmental gradients

spanning from a Mediterranean to an Alpine climate, these regions host a rich biodiversity (Médail & Quézel, 1997). They encompass a variety of land cover and land use types including large cities on the littoral, including Marseilles and Montpellier. Urban areas cover only 5% of the area (Figure S2) but the urban growth rate in recent decades has been among the highest in France. Agriculture is also largely diversified (mainly vines, fruit, vegetables and livestock) and a relatively large proportion of the production is organic (15.4% of Utilized Agricultural Land in 2014).

Defining conservation priorities

In order to define spatial conservation priorities, we used the methodology developed by Vimal et al. (2012) which compares a cumulative stressor layer (Sanderson et al., 2002), i.e. a combination of stressor layers relating to different individual human activities that impact ecosystems, with a map of biodiversity vulnerability. Both maps (cumulative stressor and biodiversity indicator) are categorized into three levels: high values (top 20%), medium values (intermediate values) and low values (bottom 40%). These two maps are combined using these levels. Highest conservation priority is given to areas with both a high biodiversity indicator and high cumulative stressor levels (classes A3, A2 and B3 in Figure 1). Low priority is given to areas with a low biodiversity indicator (B1, C1, C2 and C3, Figure 1) and intermediate priority is given to the remaining cases (A1, B2, Figure 1). The datasets used to estimate the cumulative stressor and the biodiversity indicator layers are described in Table S1.

Cumulative stressor layer

Four stressor layers were combined: transport infrastructure, urbanization, agricultural intensification, and human population density (Table 1). Stressor layers were calculated using maps with a 100 m resolution and aggregated into a grid of hexagons with 500 m sides (65 ha). For the transport stressor layer, coefficients were attributed to road types to reflect the impact different volumes of traffic might have on species (Forman 2003, Table S2), each grid

cell is then given the highest coefficient value out of all the road types intersecting that cell (BD TOPO®/RGE GIS, IGN 2008). For the urbanization stressor layer, each grid cell is given the proportion of built area (class "undifferentiated buildings" in the "built-up" data layer, BD TOPO®/RGE GIS, IGN 2008). For the agriculture stressor layer, coefficients were attributed to each of the four types of agricultural land use in the Corine Land Cover database (2006, hereafter CLC) to reflect the expected intensity of the associated practices (see Table 2). Each grid cell was then given the highest coefficient value out of the different agricultural types intersecting that cell. For the population stressor layer, each grid cell was given the value of human population density (INSEE 2010) for the predominant district in which it was found. Each stressor layer was then transformed using a Gaussian kernel smoother (within 50 km with a 25 km bandwidth for the population stressor layer and within 2 km with a 800 m bandwidth for the other three), the stressor value for each cell being equal to the sum of the values weighted by the distance-based kernel smoother. For each stressor, the values were then normalized from 0 (no threat) to 1 (highest threat). Finally, the cumulative stressor layer was obtained by summing the four individual stressor layers weighted by a coefficient of relative importance ranging from 2 to 6 (Table 1).

Biodiversity indicator

As a surrogate for biodiversity, we used a simplified indicator, based on an unweighted sum of five criteria (combined from Vimal et al., 2012 and SRCE PACA, 2013). Natural and semi-natural areas were grouped into four habitat classes: forest (all forest types), semi-open (e.g. scrublands), open (e.g. grasslands and pastures) and blue habitat (e.g. inland and maritime wetlands). The five criteria were: (i) The hydro-biological potential, relating to the biodiversity promoted by the interface of terrestrial and aquatic environments (Naiman et al., 1993) which was calculated by summing the densities of watercourses, water bodies, and wetlands; (ii) Landscape heterogeneity, commonly associated with a high level of biodiversity, which was quantified using Shannon diversity for the four habitat types (forest,

semi-open, open, blue) weighted by the total area they cover in each hexagon; (iii) The naturalness, which was calculated as a sum of the areas covered by the different types of land use (CLC Database 2006), weighted by an indicator of their naturalness (artificial: 0, discontinuous artificial: 0.1, agricultural: 0.5, natural and pastures: 1) and divided by the surface of the hexagon; (iv) The patrimonial responsibility, relating to the presence of endemic and rare species, which was calculated using the proportion of '*Zones Naturelles d'Interêt Faunistique et Floristique*' (ZNIEFF, type 1) in each hexagon; (v) The integrity of habitats, identifies contiguous interrelated natural habitats which form a cohesive unit in terms of ecological structure. In each hexagon we summed the area of habitat cover weighted by coefficients assigned based on the surface of contiguous patches (see Table S3) and by the rarity of the habitat (ratio between the habitat surface to the total surface in natural cover in the study area). We normalized (between 0 and 1) and summed these five criteria. Overall, this simplified indicator takes into account the large areas of a given type of habitat and diverse landscapes which both favor different species and different components of regional biodiversity. We considered only one single simplified descriptor of biodiversity in order to be able to test the effect of stressor estimates on the definition of spatial conservation priorities (Vimal et al., 2012).

Sensitivity analysis

In order to analyze the sensitivity of the methodology to how the cumulative stressor layer is parameterized, we compared the different maps of conservation priorities obtained using different estimates of cumulative stressors, all of which were compared to the same simplified indicator of biodiversity (Table 1, Figure 1). We tested 1) the effect of the overall methodology, by comparing the 'reference' (Vimal et al., 2012) to three other methodologies proposed in the literature for assessing human footprint, 2) the effect of the coefficients and data, by comparing the 'reference' to a number of parameter combinations where the four stressors are alternatively removed from the calculation or considered with a different

weighting or different input data, and 3) the effect of how agriculture is taken into account, both in terms of the values attributed to each agricultural land use and in terms of the data considered when calculating the layer.

Effect of the overall methodology

Cumulative stressor layers were calculated using three methodologies taken from the literature. The first two are methods used to estimate human footprint (Sanderson et al., 2002; Woolmer et al., 2008) and the third is a refinement of the ‘reference’ methodology developed as part of the implementation of green corridors in the land planning of one of the administrative regions considered in our study (SRCE PACA, 2013). These methodologies differ from the ‘reference’ in terms of the type of data they include and in the way in which the data are included (coefficients and distance weighting); these differences are highlighted in Table 1 and S3.

Effect of coefficients and input data

We ran different parameter combinations (Table 1) to assess the effect of: 1) the weighting coefficients by changing them for equal weights for the four stressor layers (‘coef 1’); 2) each stressor by systematically removing one of them (‘Without’); 3) the smoothing distances by modifying the Gaussian kernel smoother used for the transport stressor following the work of Forman & Deblinger (2000): within 800 m with a 400 m bandwidth (‘Transport 800m’); 4) the type of data taken into account when calculating the transport stressor layer by including linear information on railways and cable transportation and polygons relating to surfaces associated with roads and railway stations in addition to the linear information as in the SRCE parameter combination (‘Transport SRCE’, Table 1).

Effect of how agricultural practices are converted into a stressor layer

We tested two different aspects in relation to how agricultural practices are converted into a stressor layer. Firstly, we tested the effect of the coefficients attributed to each agricultural land use when converting land cover into a stressor layer. We reclassified CLC agricultural land use in two different ways (Table 2): according to the main crop types from the Bioscore

project ('Agri-bioscore', Overmars et al., 2014), or using our own coefficients to reclassify CLC crop types based on a literature review (Table S4) of the effects of agricultural practices on biodiversity in Mediterranean regions ('Reclass'). Secondly, we took into account complementary information on agricultural practices in order to go beyond the basic 'crop type' classification. It is well established that the positive and negative effects of agriculture on biodiversity are mediated by the intensity of practices and the presence of semi-natural elements on farmland (e.g. hedges, isolated trees), as much as by the crop types themselves (McMahon et al., 2012). We therefore used the High Nature Value index (HNV, Pointereau et al., 2007, 2010) which reflects the average intensity of agricultural practices at the municipality level, based on the proportion of hedges, pastures and non-intensive agriculture and on the diversity of crops, from low intensity (high HNV value: 30) to high intensity (low HNV value: 0). We ran two different tests in which we modified the values attributed to CLC crop types in order to reflect the HNV status, multiplying the values associated with crop types by the value associated with the HNV class defined in the Table 2. We divided the HNV values either into four classes ('HNV classes') to isolate municipalities with high scores (above 25) that host most of the Natura 2000 areas (more than 30% of agricultural land in each municipality, Pointereau et al., 2010), or into two classes ('HNV status') using the threshold of 15, previously identified as the top 25% HNV of French utilized agricultural land (Pointereau et al., 2010).

Similarity indices

In order to compare the different parameter combinations against the 'reference' parameterization (Vimal et al., 2012), we calculated the similarity between the resulting maps of conservation priorities (priorities or 'ones' being cases A3, A2, B3; Figure 1) with the Jaccard similarity index (Legendre & Legendre, 1998), given as:

$$J(\text{test}, \text{reference}) = N_{11} / (N_{11} + N_{01} + N_{10}) \quad (1)$$

, with N_{11} representing the number of cells where the test and the reference both have a value of 1, N_{01} representing the number of cells where the test is 0 and the reference is 1, N_{10} representing the number of cells where the test is 1 and the reference is 0. N_{00} represents the number of cells where the test is 0 and the reference is 0 and $N_{11}, N_{01}, N_{10}, N_{00}$ sum to the total number of cells in the maps. The Jaccard similarity index describes the level of agreement between maps: 1, the priorities are the same; 1-0.8, there is a high level of agreement; 0.8-0.6, there is a fair level of agreement; below 0.5 there is a poor level of agreement and 0, there are no overlapping priorities between maps.

All the spatial analysis were conducted using ArcMap 10.2 (Esri 2011®) and all the statistical analyses using R 2.15.2 (R Core Team, 2012).

Results

Conservation priorities in the study region

The ‘reference’ method (Vimal et al., 2012) identified top priority areas on the littoral and close to large cities such as Marseilles, Montpellier and Nice (Figure 2 – Panel A). These areas are characterized by a high level of biodiversity due to the presence of large natural areas, a high diversity of land cover types, and a high concentration in patrimonial interest zones due to the presence of endemic and rare species. Intermediate priorities were located in lowland plains and in valley floors (See Figure S1). The areas of least concern were mainly found at high altitudes where human activities are less dense, but also in man-made areas, such as those around the Rhone Valley.

Overall, the fifteen different parameter combinations we ran generally concurred with the ‘reference’ parameterization as regards the identification of the main spatial conservation priorities (Figure 2 – Panel B). All parameter combinations similarly identified as top priorities the areas that host a high-density human population, with related activities and infrastructures, and a rich diversity of species and habitats such as the hills near Marseilles

(Calanques, Figure S3 A.). Only 1.4% of the area studied was identified as top priority by all fifteen parameter combinations. All parameter combinations also similarly identified most of the areas of least concern as areas with high levels of human activity (city centers), or remote areas less affected by high-impact human activities. 80% of the entire study area was identified as being of least concern by all fifteen parameter combinations. In contrast, there was a high level of disagreement between parameter combinations when considering valleys and low altitude hinterland. In particular, the main agricultural areas such as the Camargue (Figure S3 A.) and the Durance valley or the Corbières Mountains (Figure S3 B.), near Perpignan, show contrasting results alternating between higher and lower priorities. For the entire study area, 12% was identified as top priority by less than half of the fifteen parameter combinations and 6.2% by more than half of the parameter combinations.

Sensitivity analysis

Effects of the overall methodology

The comparison of the different methodologies revealed that they strongly differed in the way in which they identified conservation priorities (Jaccard index between 0.28 and 0.7, Table 3). The SRCE parameter combination produced the most distinct outcome (Jaccard index mean value: 0.32) with all the valleys switching from least concern to top priority (Figure S3-C). These strong discrepancies between the methodologies were found in areas with intensive agricultural activities, such as the Camargue, the Rhone Valley (Figure S3-A) and east of the Corbières Mountains (Figure S3-B).

Effect of coefficients and input data

Generally speaking, the method for defining spatial conservation priorities proposed by Vimal et al. (2012), was found not to be very sensitive to the different coefficients used to weight the four stressor layers ('Coef 1', Table 3), nor to the omission of the urban stressor ('Without urban'), nor to the way the transport stressor layer was calculated ('Transport 800m', 'Transport SRCE'). It was however, more sensitive to the omission of the transport ('Without transport') and the population stressors ('Without population', Table 3). Due to its effect over

longer distances (up to 50km), the population stressor had the greatest effect on the definition of spatial conservation priorities (Jaccard 'without population': 0.59).

Effect of the way in which agricultural practices are converted into a stressor layer

The agriculture stressor was found to be important when defining spatial conservation priorities (Table 3). Omitting the agriculture stressor led to significant differences and spatial discrepancies in the identification of the conservation priorities (mean Jaccard: 0.65). While top priorities are scattered over the north of the Camargue, the Corbières Mountains and the central region between the Rhone and Durance valleys when the agriculture stressor is taken into account, they are contiguous to areas identified by both parameter combinations, in particular on the littoral and north of Montpellier, when the agriculture factor is omitted (Figure S4). The different sets of coefficients attributed to main crop types ("Agri-Bioscore" and "Reclass") also led to large differences in the identification of top conservation priorities (mean Jaccard: 0.71, Table 3). Finally, including refined data on agricultural practices ("HNV classes" and "HNV binary") led to surprisingly small changes in the priorities (mean Jaccard: 0.88 when compared to the reference, and 0.96 between them). However, these small differences are spatially clustered close to the hills on the littoral (not shown).

Discussion

How reliable is the definition of spatial conservation priorities?

Being able to reliably define spatial conservation priorities is crucial, as they can have far-reaching consequences when applied to land use and conservation planning (Wilson et al., 2009). Here we show that defining spatial conservation priorities by combining biodiversity indicator and cumulative stressor layers (Vimal et al., 2012) is generally not very sensitive to the main parameters and entry data used to estimate cumulative stressor layers. Nonetheless, we have identified two main sources of uncertainties for this method.

Firstly, our results identified two groups of stressor layers. On the one hand, some variables spatially overlap with others and their inclusion had no real impact on the definition of spatial conservation priorities (e.g. the urban stressor layer overlaps with the transport and population stressor layers). On the other hand, some variables are unique and including them or modifying any of the coefficients or input data related to them leads to major changes in conservation priorities. For instance, agriculture and high-speed transport infrastructure (highways and high-speed railways) are found in areas that are not already affected by the other stressors. Particular care should therefore be taken when mapping non-overlapping stressors, notably to ensure accurate and recent input data are used (Halpern & Fujita, 2013; Wilson et al., 2005).

Secondly, the method is highly sensitive to more extensive changes and to the general decisions taken regarding how to handle the data, what data to use and the associated distance effect (effects of the ‘overall methodology’). Modifying the reference method (‘Vimal et al., 2012’) to meet the requirements of regional stakeholders (‘SRCE PACA, 2013’) significantly modified the results of the analysis. This demonstrates that different groups of people co-constructing priority-setting with the same input data could come up with completely different assessments.

Furthermore, it is important to note that this was a partial sensitivity analysis. Many other sources of uncertainties could have been included, in addition to those tested: (i) the top priorities are here arbitrarily defined according to set thresholds (top 20%) but any other value could be used, or priorities could be defined on an area-specific basis, or according to the availability of funds; (ii) we used a simplified indicator for biodiversity, but other definitions could be used (e.g. directly including data on species, Vimal et al., 2011, or a set of different biodiversity descriptors, Vimal et al., 2012); (iii) we only considered the uncertainties related to the top conservation priorities, i.e. the areas in which direct protection actions are required,

although this method has the advantage of also distinguishing between two types of areas of ‘least concern’ where different management strategies can be applied (areas with high human pressure and low biodiversity or area with low human pressure and high biodiversity). All these uncertainties should be explicitly included in the priority-setting process and should be clearly reported to improve conservation decision-making (Burgman et al., 2005; Wilson et al., 2009).

Accounting for agriculture when defining conservation priorities

The way in which the agriculture stressor is defined is a major source of uncertainty in the method presented here. For several reasons it leads indeed to spatial shifts in the identification of top conservation priorities.

Firstly, the agriculture stressor is one of the non-overlapping stressors previously identified. Thus, whichever categories and coefficients are used to characterize agriculture, including an agriculture stressor even with a low weighting coefficient (here 3/17), leads to significant changes in the top conservation priorities.

Secondly, the way crop types are categorized into classes of management intensity (as a surrogate for human impact on biodiversity) strongly influences priority setting. Indeed, in areas where agriculture is the only human pressure, identifying crop types as more or less intensive will lead to contrasting results. The issue here is that the categorization of crop types and the coefficients allocated to these categories are highly arbitrary and case specific. Crop type intensity may indeed depend on the study region, for instance arable land in our Mediterranean study area is on average less intensively farmed than the same crop types in northern France (Agreste, 2014; Agreste PACA, 2009; Eurostat, 2015), but it is unclear how more general coefficients can be determined. By conducting our own review of the literature (Table S4), we came up with another set of coefficients, better adapted to Mediterranean crops (‘Reclass’), that also led to different top conservation priorities.

Thirdly, crop types are known to be a poor proxy for the intensity of agriculture. Indeed, what really matters is not the crop type *per se*, but the practices used for growing the crops (Andersson et al., 2013; Clough et al., 2007; Kleijn et al., 2006). In particular landscape heterogeneity and the diversity of crop types, as well as the presence of agri-environmental components such as hedges or vegetated field margins may enhance biodiversity (Bassa et al., 2011; Pino et al., 2000; Weibull & Östman, 2003). We tried to address this issue by including more refined information on the low-intensity practices (High Nature Value index for farmlands, Pointereau et al., 2007) expected to be associated with a high rate of biodiversity, in terms of species richness or habitat diversity (Aue et al., 2014; Doxa et al., 2010). However, this did not lead to significant changes in the top conservation priorities. This might be due to (i) the low spatial resolution of the HNV data (municipality level), (ii) the fact that this index is not always derived from direct information on practices but from indirect proxies such as crop type, (iii) the fact that the organic label alone is not an obvious indicator of biodiversity-friendly practices as market trends lead to the development of large-scale organic monocultures (Kremen et al., 2012) which clearly contrast with multifunctional and diversified organic farming systems relying on functional agrobiodiversity (Dib et al., 2012; Miñarro & Prida, 2013).

Taken together, these elements demonstrate the need for a better understanding of the links between agricultural practices and biodiversity. There is no doubt that the same questions about the inclusion of agriculture need to be asked of all other tools used for systematic conservation planning. Estimating the positive and negative effects of agriculture on biodiversity thus remains a challenge which needs to be met in order to develop robust conservation strategies that rely on agricultural practices to maintain open habitats, diverse landscapes and permeable areas for species from other habitat types (Mossman et al., 2015).

For the implementation of ‘green corridor’ (*Trames Vertes et Bleues*) policies in France, no strict common methodological guidelines have been proposed and the definition of spatial conservation priorities is done differently in each region. In particular, agricultural land has been considered differently in different regions and intensive arable land has been often omitted both as a potential habitat and as a stressor (DREAL Centre, 2014; ECOSCOOP, 2014). In some regions however, agricultural land has been considered as a potential corridor for species from closed or semi-open habitats (DREAL Aquitaine, 2014; Région Rhône Alpes, 2014). Indeed, these discrepancies call into question the practice of stitching together regional assessments obtained using different methods to inform the situation at national level.

Concluding remarks

Defining spatial conservation priorities requires an adaptable and flexible approach to ensure that land planning can effectively integrate new opportunities, feedback, and improved knowledge as this becomes available (Game et al., 2013). Incorporating these conservation priorities into land planning also requires the underlying uncertainties that can negatively impact the reliability of prioritization tools to be communicated. Participatory methods, such as the approach proposed by Pert et al. (2013) is an interesting way of iteratively and interactively updating spatial conservation priorities by modifying individual stressors and biodiversity surrogates according to stakeholder decisions. This could also be a helpful way of communicating on parameterization-related uncertainties.

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<Figure>Figure 1: A schematic presentation of the different steps in the calculation of

conservation priorities in Mediterranean France (Languedoc Roussillon and Provence Alpes

Cote d'Azur), following the method proposed by Vimal et al. (2012). The priority setting was

obtained from the confrontation between biodiversity indicator (in green) and cumulative stressor layer (in red).

<Figure>Figure 2 - Panel A: Conservation priorities obtained with the ‘reference’ method for the study area (Vimal et al., 2012). Black: Top priorities (A3, A2, B3 in Figure 1), dark grey: intermediate priorities (A1, B2 in Figure 1), and light grey: least concern.

<Figure>Figure 2 - Panel B: Sum of top conservation priorities identified by our 15 different parameter combinations. Black: top priority area under all the 15 parameter combinations, dark grey: priority area under more than half of the parameter combinations, medium grey: priority area under less than half of the parameter combinations, light grey: Least concern areas under all the parameter combinations.

TABLES

<Table>Table 1: List of the 15 different parameter combinations run in the sensitivity analysis. Elements that differ from the reference are displayed in italic and bold. The coefficients used to weigh the different stressor layers vary between 0 and 6.

Type of parameter combination	Name of the parameter combination	Transport calculation	Coefficient Transport	Coefficient Urban	Coefficient Population	Coefficient Agriculture	Coefficient Energy	Agriculture
Reference	Vimal et al., 2012	Max	6	6	2	3	--	reference
Effect of the overall methodology	Sanderson et al., 2002	<i>Sum</i>	<i>1</i>	<i>1</i>	<i>1</i>	<i>1</i>	--	<i>Sanderson coefficient</i>
	Woolmer et al., 2008	<i>Max</i>	<i>1</i>	<i>1</i>	<i>1</i>	<i>1</i>	--	<i>Woolmer coefficient</i>
	SRCE	<i>Sum</i>	<i>4</i>	<i>2</i>	<i>2</i>	--	<i>1</i>	--
Effect of coefficients and inputs data	Coef 1	Max	<i>1</i>	<i>1</i>	<i>1</i>	<i>1</i>	--	reference
	Without Transport	--	<i>0</i>	6	2	3	--	reference
	Without urban	Max	6	<i>0</i>	2	3	--	reference
	Without population	Max	6	6	<i>0</i>	3	--	reference
	Transport 800m	Max	6	6	2	3	--	reference
	Transport SRCE	<i>Sum</i>	<i>4</i>	6	2	3	--	reference
Effect of how agricultural	Without agriculture	Max	6	6	2	<i>0</i>	--	--

practices are converted into a stressor layer	Agri-bioscore	Max	6	6	2	3	--	<i>Bioscore coefficient</i>
	Reclass	Max	6	6	2	3	--	<i>Literature based coefficient</i>
	HNV classes	Max	6	6	2	3	--	<i>Multiplying coefficient</i>
	HNV binary	Max	6	6	2	3	--	<i>4-class value</i>

<Table>Table 2: Coefficients used to test the effect of the way in which agricultural practices are converted into a stressor layer. The upper part of the table presents the different coefficients used to convert agricultural land use from Corine Land Cover (CLC) into an agriculture stressor layer under different scenarios. The lower part gives the different multiplying coefficients associated with levels of High Nature Value (HNV) index (Pointereau et al., 2007).

Crop types CLC level 2	Crop types CLC level 3	Code	Vimal et al., 2012	Sanderson et al., 2002	Agri- bioscore	Reclass
Arable Land	Non-irrigated arable land	12	1	0.7	0.6	0.7
	Permanently irrigated land	13				
	Rice fields	14				
Permanent crops	Vineyard	15	0.40	0.7	0.7	0.4
	Fruit trees and berry plantations	16				
	Olive groves	17				
Pastures		18	0.15		0.6	
Heterogeneous agricultural areas	Annual crops associated with permanent crops	19	0.65	0.6	0.4	0.3
	Complex cultivation patterns	20				
	Land principally occupied by agriculture, with significant areas of natural vegetation	21				
	Agro-forestry areas	22				
Test name		Value of HNV		Coefficient		
HNV classes		> 25		0.25		
		20 - 25		0.5		
		15 - 20		0.75		
		< 15		1		
HNV binary		> 15 (HNV status)		0.5		
		< 15 (No HNV status)		1		

<Table>Table 3: Comparison between the conservation priorities obtained under the different tests (Jaccard's similarity index). In the last part about the agriculture stressor layer the color identify different types of comparisons: in green the effect of omitting the agriculture stressor layer, in red the effect of the different coefficient sets assigned to crop types, in black the effect of taking agricultural practices into account on top of crop types, and in blue the comparison between two ways to account for HNV (High Natural Value farming) data. In brackets are the comparison that involve tests differing by more than one element.

Effect of the overall methodology	Reference (Vimal et al., 2012)	Sanderson et al., 2002	Woolmer et al., 2008	SRCE
Reference (Vimal et al., 2012)	-			
Sanderson et al., 2002	0.46	-		
Woolmer et al., 2008	0.27	0.28	-	
SRCE	0.35	0.30	0.31	-

Effect of coefficients and inputs data	Coef 1	Without transport	Without urban	Without population	Transport 800m	Transport SRCE
Reference (Vimal et al., 2012)	0.70	0.61	0.94	0.59	0.78	0.92

Effect of how agricultural practices are converted into a stressor layer	Reference (Vimal et al., 2012)	Without agriculture	Agri-Bioscore	Reclass	HNV classes	HNV binary
Reference (Vimal et al., 2012)	-					
Without agriculture	0.64	-				
Agri-Bioscore	0.79	0.56	-			
Reclass	0.68	0.68	0.66	-		
HNV classes	0.88	0.68	(0.75)	(0.7)	-	
HNV binary	0.87	0.69	(0.74)	(0.71)	0.96	-

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