Integrated methods for monitoring the invasive potential and management of *Heracleum mantegazzianum* (giant hogweed) in Switzerland

Ross T. Shackleton[1*], Blaise Petitpierre[2], Mila Pajkovic[2], Florian Dessimoz[2], Olivier Brönnimann[2], Loïc Cattin[1], Šárka Čejková[3,4], Christian A. Kull[1], Jan Pergl[3], Petr Pyšek[3,4], Nigel Yoccoz[5] and Antoine Guisan[2]

[1]Institute of Geography and Sustainability, University of Lausanne, 1015 Lausanne, Switzerland

[2]Department of Ecology and Evolution (DEE), University of Lausanne, Biophore, CH-1015, Lausanne, Switzerland

[3]Institute of Botany, Department of Invasion Ecology, The Czech Academy of Sciences, CZ-252 43 Průhonice, Czech Republic;

[4]Department of Ecology, Faculty of Science, Charles University, Viničná 7, CZ-128 44 Prague, Czech Republic

5 Department of Arctic and Marine Biology, UiT The Arctic University of Norway, N-9037 Tromsø, Norway

*(rtshackleton@gmail.com)

Abstract

Biological invasions are a major driver of human induced global environmental change. This makes monitoring of potential spread, population changes and control measures necessary for guiding management. We illustrate the value of integrated methods (Species Distribution Modelling (SDM), plant population monitoring and questionnaires) for monitoring and assessing invasions of *Heracleum mantegazzianum* (giant hogweed) over time in Switzerland. SDMs highlighted the potential spread of the species, uncovered ecological mechanisms underlying invasions, and guided monitoring at a regional level. We used adaptive and repeat plant sampling to monitor invasive populations status and changes and assess the effectiveness of *H. mantegazzianum* management over three periods (2005, 2013 and 2018) within the pre-Alps, Vaud. We also conducted questionnaire surveys with managers and the public. Multi-scale modelling, integrating global and regional SDMs, provided the best predictions, showing that *H. mantegazzianum* can potentially invade large parts of Switzerland, especially below 2000 m.a.s.l. Over time populations of invasive *H. mantegazzianum* in the Vaud pre-Alps have declined, which is most likely due to a sharp rise in management uptake post 2007 (7 % of municipalities before 2007 to 86 % in 2018). The level of known invasive populations have decreased by 54 % over time. Some municipalities have even successfully eradicated *H. mantegazzianum* within their borders. However, a few areas, particularly in the rural, higher altitude municipalities, where management was not implemented effectively, populations have expanded, which could hamper control efforts at lower altitudes. We provide

encouraging evidence that control measures can be effective in reducing plant invasions with long-term commitment as well as a good template for using integrated methodological approaches to better study and monitor invasive alien species.

Keywords:

Biological invasions, Bioclimatic modelling; environmental management; invasive species; monitoring, invasive plants

1. Introduction

Biological invasions are among the leading human-induced drivers of global environmental change, resulting in negative effects on biodiversity, ecosystem services, human well-being and livelihoods and can result in socio-ecological regime shifts (Vilà et al. 2011; Pyšek et al. 2012; Jeschke et al. 2014; Shackleton et al. 2018). They arise from the purposeful or accidental movement of species outside their native ranges to new locations, whereby through a number of mechanisms they are able to spread over wide areas (Blackburn et al. 2011). Globally, introductions of invasive alien species (IAS) have not reached a saturation point and their threats are still increasing (Seebens et al. 2017). Furthermore, many established IAS continue to spread rapidly in their introduced landscapes resulting in negative impacts (Shiferaw et al. 2019). Due to the negative impacts of IAS on humans and the environment, it is important that they are efficiently managed, in terms of costs and objectives. However, it needs to be noted that not all IAS pose negative impacts, and for those that need to be managed, the approaches should take the environmental and socioeconomic context into account to develop appropriate and optimal management strategies (Kull et al. 2011; Pergl et al. 2016; Bach et al. 2019). For example, species with high negative social and ecological impacts, low benefits and those that can be managed cost effectively should be prioritised (van Wilgen et al. 2012).

A number of management options are available for IAS, with their suitability depending on the invasive species traits, local social and environmental settings and their position on the introduction-naturalisation-invasion continuum (Blackburn et al. 2011; Wilson et al. 2011; Bach et al. 2019). Key parts of managing invasive species include monitoring the current state of invasion, monitoring management implementation effectiveness as well as anticipating further spread (Blossey 1999; Maxwell et al. 2009; Downey 2010; Shackleton et al. 2017). Effective monitoring is lacking for IAS management in many areas and for countless other environmental-related programs, representing a major barrier to efficient environmental management (Pergl et al. 2012; Shackleton et al. 2016; Turner et al. 2016; van Wilgen et al. 2016). Monitoring is key for assessing the effectiveness of different environmental control actions over time (Yoccoz et al. 2001), and can help guide relevant adaptations of management strategies. Various tools are available for monitoring IAS, and can be combined to provide more holistic understanding. Options for monitoring and mapping different IAS vary depending on the spatial scale, and the target species' characteristics. For example, remote sensing and Google Earth can be used to monitor species that are prominent in the landscape over wide spatial scales (Kennedy et al. 2009; Müllerová, et al. 2013; Van den Berg et al. 2013; Visser et al. 2014). Vegetation monitoring along roads can also be used to record coarse-scale distributions of IAS and is particularly useful when there is little knowledge on invasions in the region (Henderson 2007; Rejmánek et al. 2017; Witt et al. 2019). Plant populations or individual plants can be monitored with GPS locations at localised scales, which is vital for early detection and subsequent rapid management response (Panetta 2006; Kaplan et al. 2012). To predict the potential distributions of IAS, environmental niche-based Species Distribution Models (SDMs) (Guisan and Thuiller 2005; Elith and Leathwick 2009) are increasingly used in management and risk assessment studies (Peterson 2003; Thuiller et al. 2005; Vicente et al. 2011; Guisan et al. 2013). SDMs can be used for (i) guiding the finding of populations that need to be managed, (ii) highlighting risks of future spread, and (iii) better understanding ecological factors underlying distributions and spread (Guisan and Thuiller 2005; Vicente et al. 2016). Targeted interviews or surveys or analysis of reports can be used to gather information regarding cost and management invitations conducted by the authorities, practitioners or the public (Shackleton et al. 2015; van Wilgen et al. 2016). Lastly, participatory approaches can also be used, whereby people help to monitor invasion and provide information relating to population changes and management effectiveness and can link to citizen science and volunteering (Bryce et al. 2011; Adriaens et al. 2015; Mohanty and Measey 2018; Pagés et al. 2019), a tool also common in other environmental management and conservation projects.

One invasive species that is present across large parts of the northern hemisphere and poses threats to humans and the environment, and so needs to be managed is *Heracleum mantegazzianum* Sommier & Levier (giant hogweed in English, and la Berce du Caucase in French) (Pyšek et al. 2007). *Heracleum mantegazzianum* is a monocarpic perennial forb of the Apiaceae family and is a widespread IAS globally, being particularly prominent in many parts of Europe but also present and starting to spread rapidly in North America (Nielsen et al. 2005; Page et al. 2006; Pyšek et al. 2007; Pyšek et al. 2008)., Native to the southern side of the Western Greater Caucasus in Russia and Georgia, where it grows in species-rich tall-herb mountain meadows, clearings and forest margins up to the treeline of ~2000 m a.s.l., it was first introduced to Europe as an ornamental to Kew Gardens, UK, in 1817 (Jahodová et al. 2007) from which seed was spread to other gardens in the UK and Europe. From these planted sites *H. mantegazzianum* has escaped and invaded natural areas in at least 19 European countries in 14 of which it was first recorded before 1900 (Pyšek et al. 2008; Herry et al. 2009). It now invades primarily along meadows and water courses, where it can cause increased erosion along riverbanks (Trottier et al. 2017; Moravcová et al. 2018). *Heracleum mantegazzianum* also has ability to produce vast number of seeds, regenerate, tolerate disturbances and high competitiveness making it a common and persistent invasive (Pyšek et al. 2007). The species forms dense mono-specific stands therefore reducing native species diversity in invaded areas (Thiele and Otte 2007; Hejda et al. 2009; Jandová et al. 2014; Moravcová et al. 2018). It is also dangerous for human health, as the sap contains furanocoumarins, which lead to serious skin burns (Lagey et al. 1995). In Switzerland, *Heracleum mantegazzianum* was first introduced into Geneva in 1895 and seed was later transported throughout the country into alpine botanical gardens and later private gardens in the early 1900s (Jeanmonod 1999; Dessimoz 2006). The species is common in western Swiss Alps where climatic conditions are similar to its native rage in the Caucasus (Henry et al. 2009). Due it spread and impacts to humans and the environment, *H. mantegazzianum,* is now on Switzerland's Black List of IAS (www.infoflora.ch) and considered to be among most threatening and worst IAS in the country.

In this paper we used integrated methods to better understand and monitor the distribution, population dynamics and management effectiveness of *H. mantegazzianum* in Switzerland. This included: (i) building SDMs, using a multi-scale approach (Gallien et al. 2012; Petitpierre et al. 2016), to understand potential distributions and guide population level sampling, (ii) using adaptive sampling to try and estimate total population sizes in the study region (Thompson and Seber 1996; Thompson 2012), and fixed-point population monitoring to specifically assess population changes of *H. mantegazzianum* between 2005 and 2018, and (iii) conducing questionnaires with local municipalities (communes in Switzerland) and the public regarding *H. mantegazzianum* threats and control. This work should provide guidance for the control of *H. mantegazzianum* in the study region but can also be used as a template to guide the future study and monitoring of other IAS in different regions of the world using mixed method approaches (Federal Office for the Environment 2006).

2. Methods

2.1 Study area

Modelling was conducted at two scales; at a global level and projected for Switzerland and a second regional level model calibrated for Switzerland was projected for the pre-Alps in Vaud canton, (Figure 1 and 2). Field work was conducted in the pre-Alps area which is located between the Rhône Valley, and the south-west edge of the high Alps and covers 564 square km area. Elevations range from 372–3 210 m.a.s.l. and the dominant bedrock is calcareous. Annual mean temperatures range from -3 to 10°C, depending on elevation, while mean total precipitation ranges from 1 060–2 400 mm per year (Randin et al. 2006; Henry et al. 2009). Winters are cold and wet, with abundant snowfall. The region is relatively densely settled, with rural land use focused on forestry, dairy farming, and vegetable and fruit agriculture (at lower elevations), as well as on winter and summer tourism (snow sports, hiking). The study area is covered by 25 municipalities (communes), with responsibilities for local land management.

2.2.1 Hierarchical Species distribution modelling

A multi-scale modelling approach was used (Gallien et al. 2012; Petitpierre et al. 2016). This included a global model approach fitted to Switzerland and a further refined regional model fitted to the pre-Alps of Vaud. For species where the equilibrium assumption does not hold (i.e. the species is not in equilibrium with its environment, as is the case for IAS), using the output of global models significantly improves the predictive power of finer scale regional models (Gallien et al. 2012).

2.2.1.1 Approach for the Global Species Distribution Model

For the global model approach species occurrences were used from the widest possible range of *H. mantegazzianum*. Coordinates of species occurrences were extracted from the GBIF database (www.gbif.org), providing data points mostly for Central and Western Europe and North America (14 047 points total), and from Info Flora (the Swiss national floristic database: www.infoflora.ch) (2 978 points). For the native range, a further 11 population coordinates were taken from Henry et al. (2009), and a further 42 population coordinates were provided by co-authors from their personal records. This model therefore included data from both the native and invaded range to include the widest possible niche, thus improving predictive power over large scales (Broennimann and Guisan 2008).

Only occurrences with a precision higher than 1 500 m were kept, leading to the inclusion of 9 813 occurrences in total. As the occurrence points were aggregated, occurrences were selected randomly within each aggregate, by setting a 10 km minimal distance between occurrences "occurrence thinning", thus reducing the effect of occurrence clusters (Verbruggen et al. 2013). This resulted in 1 617 occurrences after disaggregation. As the delimitation of the study area used to calibrate SDMs can have an important impact on predictions (Barve et al. 2011), we tested three different calibration backgrounds as extents for selecting the pseudoabsences, using: i) the whole world, ii) biomes (Olson et al. 2001), and, iii) ecoregions (Olson et al. 2001) layers. Within each of these three extents, 10 000 pseudoabsences were randomly sampled. The three model outcomes were compared and the one with the strongest statistical support was chosen.

We primarily considered climatic variables for the global model, as they have the most important influence at large scales (Woodward 1987; Thuiller et al. 2004). The 19 bioclimatic variables from Hijmans et al. (2005) and soil water balance variable (www.cgiar-csi.org) were considered at a 30 Arc seconds (about 1 km) resolution (See Supplementary material Appendix 1). In order to select the best predictors for the global model and to avoid high correlation between predictors, the initial 20 predictors were clustered based on their correlation after extraction for each of the model calibration extents, and grouped into nine equidistant clusters (Supplementary material, Appendix 1). One predictor in each group was then selected (from the ecoregions correlation clusters), based on it having the most direct ecological effect on the study species (Guisan and Zimmermann 2000; Petitpierre et al. 2017: see Supplementary Material, Appendix 1).

This global SDM based on the climatic niche was used in two ways: (i) to predict the species' potential distribution at large scales (Switzerland), and, (ii) to further weigh pseudoabsences towards areas predicted as suitable, to be used in the regional models' development.

2.2.1.2 Approach for the Regional Species Distribution Model

The regional model considered information from the global model approach (in the form of weighted pseudo-absences), but was also calibrated using topographic and anthropogenic variables, in addition to climatic ones (see Supplementary material, Appendix 1, Table 1). Such variables affect species distributions at finer scales more than climate (Mod et al. 2016; Petitpierre et al. 2016).

For the regional models, species occurrences for Switzerland were obtained from Info Flora (see above), and populations with a precision higher than 100 m were included. All data points recorded by Dessimoz (2006) were also included. This resulted in a total of 2 361 occurrence points for Switzerland. Occurrences in Switzerland were disaggregated, keeping a minimum distance of 250 m between them to avoid spatial autocorrelation, resulting in the inclusion of 1 304 occurrence points. Two sets of 10 000 pseudoabsences were generated for the regional scale (Switzerland). A first set, to be used for model calibration, was biased towards areas in Switzerland predicted as unsuitable by the global ecoregions model (i.e. more pseudoabsences in unsuitable areas (Chefaoui and Lobo 2008; Gallien et al. 2012)). A second pseudoabsence set for Switzerland was generated randomly, to be used for model evaluation. Both pseudoabsence sets were sampled across all of Switzerland, but after exclusion of altitudes over 2 500 m.a.s.l. (above which the species does not occur in Switzerland), as well as unsuitable primary surface categories such as lakes, glaciers, rock and scree (obtained from www.swisstopo.ch). For predictor selection, the same method as for the global model was used for the regional model. In total, 12 predictors were used at this scale, at a 25 m resolution, out of an initial set of 15 predictors (Supplementary material, Appendix 1 Table 1). In addition to climatic predictors, topographic and anthropogenic variables that influence the distribution of *H. mantegazzianum* were included.

This model was projected at the scale of the Vaud pre-Alps region (incorporating 25 municipalities of the canton) at a pixel resolution of 25 m in order to obtain the predicted suitable areas for *H. mantegazzianum* at a very fine scale.

2.2.1.3 Model statistical analysis and spatial projections

Both models were developed in R CRAN (R Core Team, 2012), using the biomod2 package (Thuiller et al. 2009), and fitted using three techniques: Generalized Linear Model (GLM, Guisan et al. 2002), Generalized Boosted Model (GBM, Elith et al. 2008), and Maximum Entropy model (MAXENT, Phillips and Dudik 2008). Model predictions and evaluations were then averaged into a single ensemble model (Araújo and New, 2007), in which all three model techniques were given the same weight. This approach accounts for uncertainty of individual models and leads to improvement of predictions compared to using a single modelling technique (Marmion et al. 2009; Thuiller et al. 2009). Biomod2 also assessed the importance of each predictor variable through permutations, and provided response curves of the species for each variable and modelling technique (Table 1; Supplementary material Appendix 1).

Models were evaluated using the Area Under the receiver operating characteristic Curve (AUC) (Fielding and Bell 1997) and a maximisation of the True Skill Statistics (TSS; Allouche et al. 2006; i.e. maxTSS; see Guisan et al. 2017) (Table 1; Supplementary material Appendix 1). These two indices include both presences and absences in the evaluation. As biological invasions are ongoing processes and all suitable areas may not be colonized, we also computed the continuous Boyce index evaluating how much presences are discriminated from the background in the study area (Hirzel et al. 2006). Spatial projections were mapped over the study area using ArcGIS (ESRI). The whole procedure (pseudoabsence sampling, model calibration, evaluation and projection) was replicated ten times and values (for model evaluation, variable importance and suitability) were averaged across the ten replicates. Mean suitability predictions of the global ecoregions model were converted into binary predictions (suitable or unsuitable) using the threshold corresponding to the maximum TSS (Freeman and Moisen 2008), in order to investigate the distribution of suitable pixels across the elevation gradient in Switzerland.

2.2.2 Adaptive sampling and density estimation

Random-stratified adaptive sampling was conducted in suitable areas in the Vaud pre-Alps (Thompson and Seber 1996). This approach allows for the estimation of the species density and therefore of the total number of individuals in the study area (Thompson, 2012). It is ideal for sparse but highly clustered species, as is the case for *H. mantegazzianum*, which occurs in dense stands (Tiley et al. 1996). The sites visited were chosen based on a random-stratified design: the regional SDMs' continuous suitability was reclassified into ten strata in the Vaud pre-Alps study area, after exclusion of unsuitable primary surface categories. In each stratum, ten points were randomly chosen, resulting in 100 sites to be visited (25 m × 25 m plots). If the species was present in one of the sites, the adaptive sampling method was carried out - whereby four neighbouring plots were equally sampled, and the procedure repeated until the species was no longer found in neighbouring plots, resulting in a network of plots that represents the whole population cluster (see Supplementary Material, Appendix 2). For each visited site, we recorded a description of the site, as well as presence or absence of the study species, and if present, the number of individuals, their percentage of surface cover, and the presence of flowering individuals. Estimation of the actual *H. mantegazzianum* population density was carried out following the methods of Thompson and Seber (1996) (see Supplementary material, appendix 2, text and equations). This was carried out in 2005, 2013 and in 2018, in order to assess the change in invasion status over a 13 year period. Sampling was done in mid-late summer each time to allow for population stabilisation and comparability of population status.

2.2.3 Fixed population monitoring

In total, 51 *H. mantegazzianum* populations whose locations were taken from InfoFlora (www.infoflora.ch) and their presence verified during field work in 2005, were revisited in 2013 and 2018. At each site, we recorded the population status, this included: if the population was present, we assessed the change in population size (recorded number of plants, information of the population (i.e. presence of flowering plants, all juveniles etc.) and if patch size increased, decreased or remained stable between monitoring times. Furthermore, signs of any management or disturbance were also recorded if they were visible (i.e. evidence or mowing, cutting or herbicide application or indirectly through land use changes). Chi-squared tests were used to compare the persistence and change of the populations between the three time periods.

2.2.5 Outreach and questionnaires

2.2.5.1 Assessing management activities for Heracleum mantegazzianum with management officials

In 2006 all municipalities in the Vaud pre-Alps (25) received the results of the first population monitoring campaign and the SDM model-based estimates of invasion potential (i.e. map of suitable habitats for the species in their region) (Dessimoz 2006). This was done to raise awareness and for helping municipalities to better target and co-ordinate management efforts for controlling of *H. mantegazzianum* invasions.

The same 25 municipalities were again contacted in 2013 and 2018, with a questionnaire to find out whether management efforts had been carried out, and if so what kind of measures and how often they had been done. It also included open ended questions relating to perceptions of management success and failure.

2.2.5.2 Assessing the perceptions of impact and management by farmers and the general public

During 2018, using a semi-structured questionnaire (in French), we interviewed 69 people including 58 representatives of the general public and 11 farmers in the Vaud pre-Alps region. Farmers were approached and contacted though agricultural lists, snowballing and through the use of door-to-door surveys. For the general public questionnaires were sent out to locals' residence in order for them to respond. The semi-structured questionnaires had questions relating to people's knowledge of the plant, their perceptions of the plant, their views on the effectiveness of management implementation and perceptions of management (see Supplementary material, Appendix 3).

3. Results

3.1. Predictive species distribution models

3.1.1 Suitability predictions for Heracleum mantegazzianum

The suitability predictions of the global models, projected across Switzerland, were more refined for the model calibrated at the ecoregions level over those projected at larger global extents (world and biomes) (Table 1). After conversion into binary suitability, predictions of the global ecoregions were used to model suitability for Switzerland. More than two thirds (68%) of the country's surface were predicted as climatically suitable (Figure 1). Unsuitable areas were primarily around the high peaks of the Central Alps with elevations greater than 2 000 m.a.s.l. Highest suitability occurred at around 400–700 m.a.s.l. (lower floodplains and hills), with occurrence suitability reaching up to 1 900 m.a.s.l. This also holds for the Vaud pre-Alps area of Switzerland, where the highest peaks were predicted as unsuitable (Figure 2).

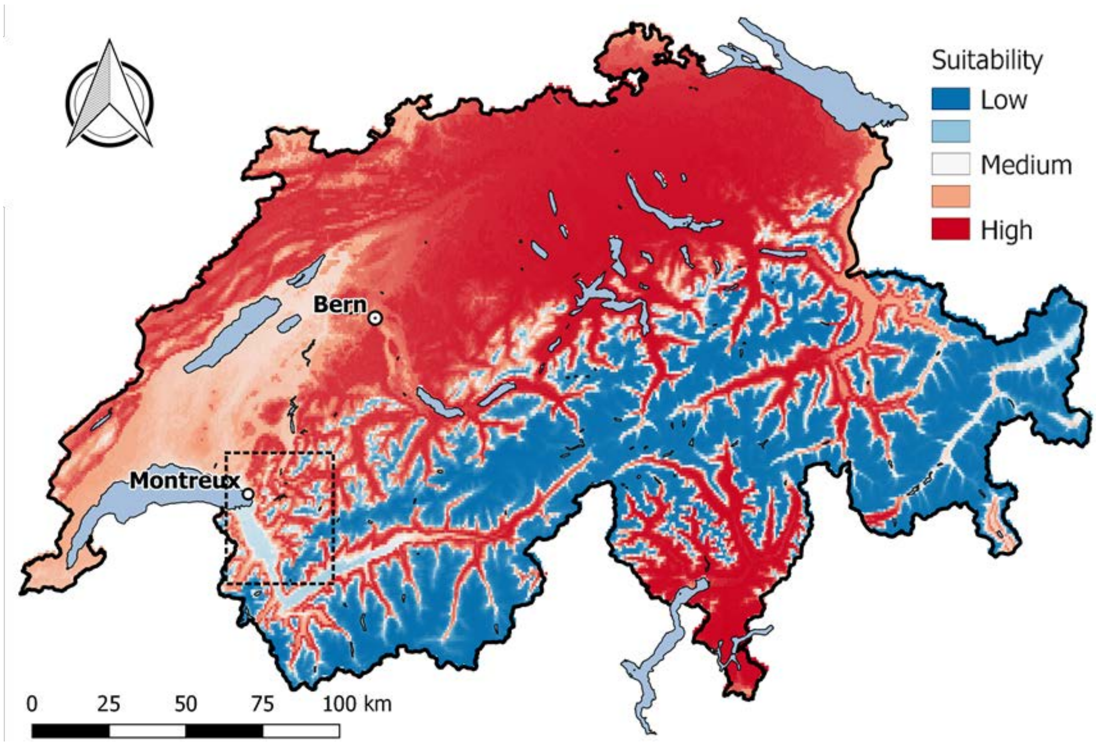

Fig 1 Suitability of *H. mantegazzianum* for Switzerland using suitability predictions of the global ecoregions distribution model (calibrated at the global scale and projected for Switzerland). The dashed rectangle represents the extent of the study area

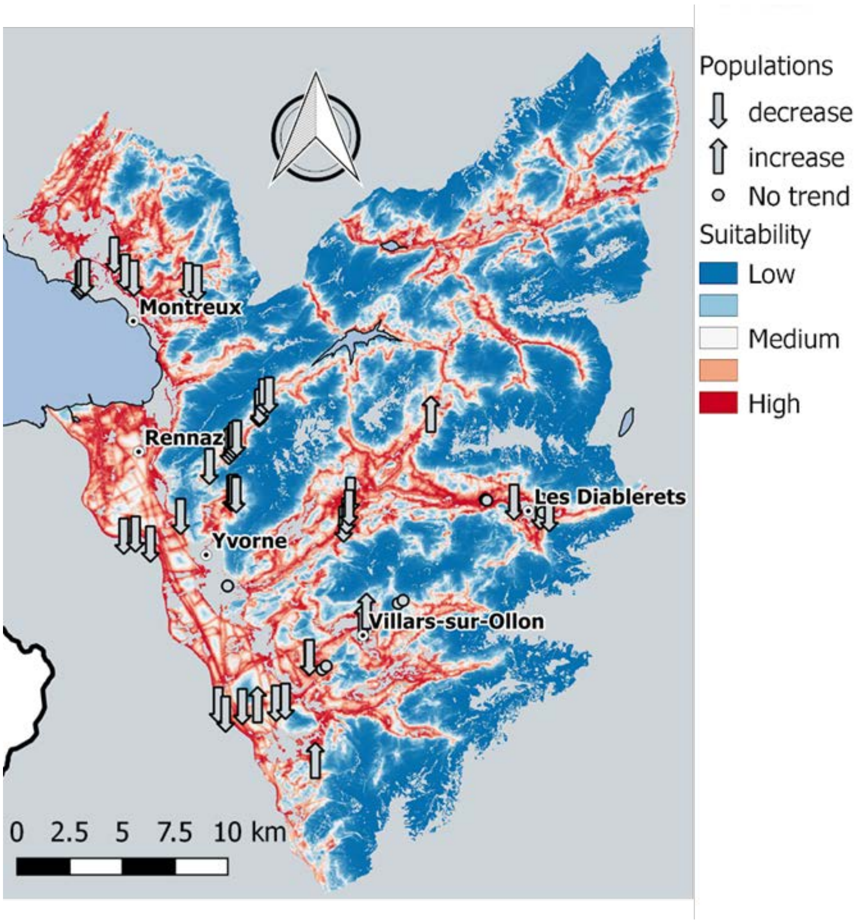

Fig 2 Suitability predictions of the regional distribution model for *H. mantegazzianum* (calibrated at the scale of Switzerland, and projected across the study area of the Vaud pre-Alps of Switzerland). Also showing changes in population clusters of *H. mantegazzianum* monitored between 2006 and 2018

3.1.2 Variable importance

The most important variables for the global models were relatively consistent among the three different model types fitted. We, however, chose to use the one based on the ecoregions extent over those based on world and biomes extents as it provided better statistical evaluations (Figure 3a, Table 1). Maximum temperature of the warmest month (bio5), minimum temperature of the coldest month (bio6), and temperature seasonality (bio4) were consistently among the three most important variables for the global models (Figure 2a, also See Supplementary material, Appendix 1). For two of the global models (those fitted at ecoregions and biome scales), yearly soil water balance (sws2) also came out as an important predictor variable (Figure 2a, Supplementary material, Appendix 1). Response curves for the global model indicate that *H. mantegazzianum* distribution is limited by extreme maximum and minimum temperatures, extreme temperature seasonality, and high aridity (i.e. low soil water balance) (Supplementary material, Appendix 1).

For the regional model conducted for the Vaud pre-Alps, the most important variable was distance to roads, followed by number of frost days during growing season (sfroyy), annual temperature standard deviation (tvar) and number of precipitation days during growing season (pday) (Figure 3b, Supplementary material, Appendix 1). Response curves for regional predictors indicate that the species is found close to roads and in areas of high temperature seasonality (Supplementary material, Appendix 1).

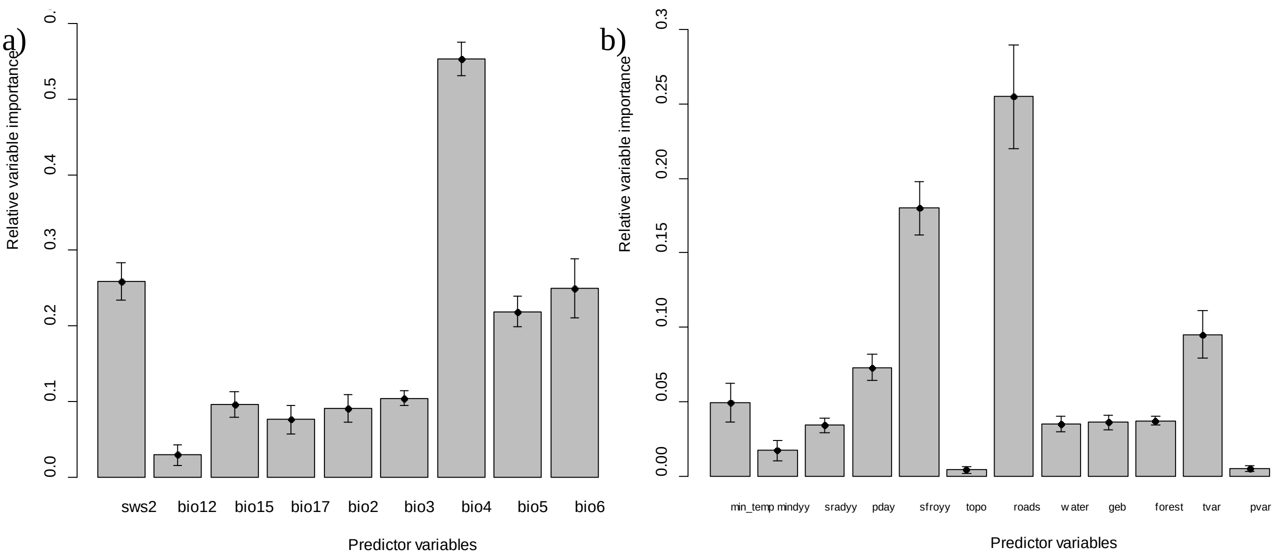

Fig 3 Variables used in the models**:** a) Importance of climatic variables included in the global ecoregions' distribution model for *H. mantegazzianum*. b) Importance of predictor variables included in the regional distribution model for *H.*

mantegazzianum. For a): Sws2 – yearly soil water balance; bio12 – annual precip.; bio 15 – precip. seasonality; bio 17 precip. driest quarter; bio2 – mean diurnal range; bio3 – isothermality; bio4 – temp. seasonality; bio5 – max. temp. warmest month; bio 6 – min. temp coldest month. For b): min_temp - annual mean (AM) of monthly mean of minimum temp., mindyy - AM of monthly moisture index; sradyy - AM of monthly global potential shortwave radiation; pday - AM number of precip. days/growing season; sfroyy - AM number of frost days during growing season; topo – topographic position; roads – Euclidean (E) distance to roads and railways; water - E distance to water; geb – buildings density; forest, E dist. forest edge; tvar - Annual sd of monthly mean of average temperature; pvar - Annual sd of monthly mean precipitation *For a description of variables corresponding to abbreviations, see Supplementary Material Appendix 1, Table 2

3.1.3 Model evaluation

The models had high evaluation values, both at global and regional scales with AUC between 0.926 and 0.991, maxTSS between 0.725 and 0.928 and Boyce index values between 0.882 and 0.967 (Table 1).
Using larger calibration extents for the global model (world or biomes) produced higher AUC and maxTSS values but lower Boyce index values than for the ecoregions model. Indeed, the world model had the highest AUC and maxTSS values, but the lowest AUC threshold, maxTSS threshold and Boyce index values (Table 1). The ecoregions model on the other hand, had the lowest AUC and maxTSS values, but the highest Boyce index (Table 1).

Table 1 Evaluation values for the global distribution models for *H. mantegazzianum* (calibrated at world, biomes and ecoregions scales, respectively), and for the regional model (calibrated at the scale of Switzerland). Values are means across ten model replicates.

Evaluation metric	**Global model approaches for Switzerland**			**Regional model for the Vaud pre-Alps**
	World	*Biome*	*Ecoregion*	
maxTSS	0.928	0.894	0.725	0.790
AUC	0.991	0.983	0.926	0.960
maxTSS threshold	437.6	483.1	481.5	531.0
AUC threshold	434.7	485.8	480.1	531.5
Boyce	0.882	0.905	0.967	0.914

3.2 Plant population monitoring

3.2.1 Stratified adaptive sampling and density estimation

2005

The stratified adaptive sampling resulted in six occurrence sites for *H. mantegazzianum* in 2005, out of the 100 sites visited. Resulting networks were composed of 4 to 24 squares, and abundances ranged from 1 to 355 plants per invasive population. This yielded an estimation of $\hat{u}_{st}$ x N = 3 300 000 plants in the study area (Vaud pre-Alps), with a variance of 8.6 %, corresponding to 29 000 individuals ($\hat{u}_{st}$ is the mean number of *H. mantegazzianum* individuals per square pixel, and N is the total number of pixels (25 m × 25 m plots) in the study area) (See appendix 2 for calculations formulas based on Thompson and Seber (1996)).

2013

Eight years later, in 2013, only one occurrence site for *H. mantegazzianum* was found among the 100 randomly visited sites. The resulting network, located at the Villars Golf Course (1 600 m. a.s.l.), in the commune of Ollon, was composed of 44 plot squares in five probability strata, and a total of 6 570 individuals. The density estimation based on this adaptively sampled population yielded an estimation of $\hat{u}_{st}$ x N = 950 000 individuals in the whole study area. Given that only one occurrence point was sampled, variance could not be calculated.

2018

Despite finding seven new populations nearby while walking to the randomised sampling points, no population of *H. mantegazzianum* was found at the 25 x 25 m plots at the 100 randomly selected points and therefore we could not conduct any calculations for 2018.

3.2.2 Population level monitoring and change

In 2005, 93 *H. mantegazzianum* occurrences were confirmed in the Vaud pre-Alps. These confirmations were based on visiting existing record points and confirming populations were present and through adding new populations that were found during field work. Most populations were found in or around the alpine resort villages of Villars-sur-Ollon and Les Diablerets. Several large populations were also established close to Lake Leman/Geneva in the region of the lac de l'Hongrin, in Nermont, above Montreux, and in Yvorne within the Rhône Valley.

In 2013 and 2018, we visited 52 of the known *H. mantegazzianum* sites recorded and confirmed in 2005. *Heracleum mantegazzianum* was found at 42 sites (81%) in 2013 and at 24 (46 %) of sites in 2018. This represents a decrease in invasive populations of 54 % from 2005 to 2018. Between 2005–2013, *H. mantegazzianum* abundance had decreased at 20 sites, was similar to that of 2005 at 14 sites, and had increased at eight sites. In terms of differences between 2013 and 2018, at 10 sites populations had decreased, 11 sites had similar population sizes and at three sites populations sizes had increased. In addition, 15 new populations were come across randomly during field work were recorded in 2013 and seven in 2018 – but were never revisited. Between 2005 and 2018 population sizes differed significantly in the study area (χ^2 = 105.5; p < 0.0001).

In 2013, 15 of the 42 invasive *H. mantegazzianum* populations had evidence of management. Most had been mown (13), while the flowering heads of two populations has been cut off and one population had evidence of herbicide application. In 2018, nine of the 24 invasive populations found had signs of management intervention. Of this most (five) had been mowed, followed by three areas where the flowering stems of adult plants had been cut. One population was also found in new crop field and larger plants had been broken presumably accidently through agricultural activity.

3.3 Management implementation reporting

The majority (21 out of 25) municipalities in Vaud responded to the 2013 questionnaire – of which two did not provide adequate information on *H. mantegazzianum* management and were thus not included in the analysis. Furthermore, another two municipalities did not have populations of *H. mantegazzianum* recorded and so could not answer. Therefore, we assessed management trends for 17 municipalities between 2005 and 2013. In the follow up survey for 2018, 23 out of 25 municipalities responded and six of them did not have any *H. mantegazzianum* recorded in their boundaries.

Only one municipality was managing *H. mantegazzianum* before 2006 and with increased awareness (awareness campaign and outreach in 2006) there has been a big increase in management over time (Figure 4). Different methods and approaches are used in the different municipalities and responsibilities of who managed and monitored invasions also varied. Methods ranged from applying herbicides, to mowing, to using machinery to remove invasive populations. Two municipalities that did not have *H. mantegazzianum* reported monitoring for it regularly. Most municipalities see management in a positive light, suggesting that management has been effective in most instances: "*sporadic outbreaks are easily removed*", "*populations are contained and in decline*". A number of municipalities have had local eradications; for example, Blonay found *H. mantegazzianum* populations in 2013 and managed them through to the end of 2015, after which the *H. mantegazzianum* was no longer present in the commune, although spread from elsewhere is possible. Similarly, Roche managed populations early on and *H. mantegazzianum* has not been recorded or managed again in the commune since 2008. Yet in other municipalities invasions are emerging; for example, in Rennaz management interventions had to be implemented for the first time in 2017 in response to new records.

A few municipalities also report facing issues with regard to management: "*we do the minimum, there is already a lack of financial means and especially staff*", "*we can only intervene in the sensitive zones near neighbourhoods, around* *housing, parks, sidewalks, pedestrian paths*" and "*there is lack of financial means allocated by the Canton of Vaud for the gigantic project; this means that the problem of invasive plants cannot be resolved by the municipalities without cantonal (provincial) help*".

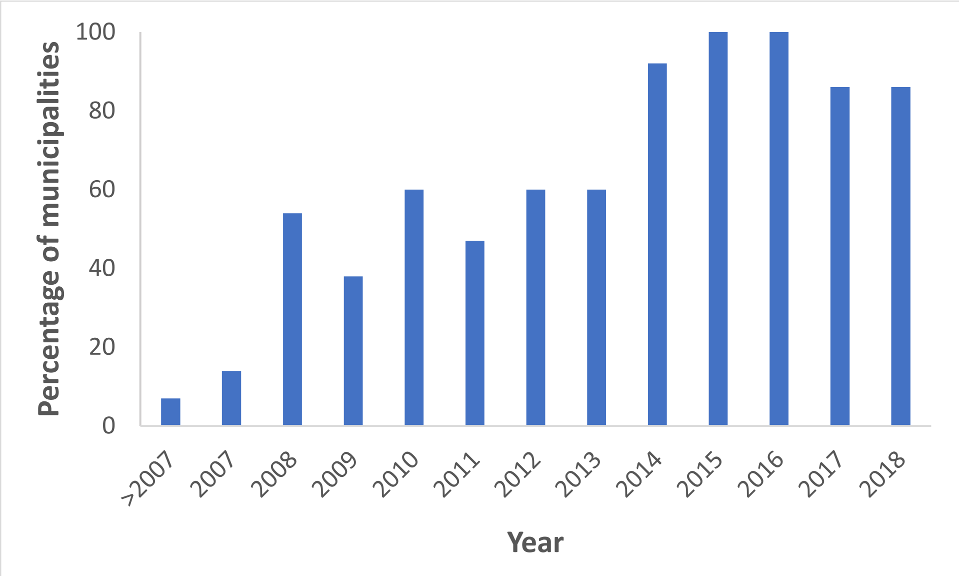

Fig 4 Percentage of municipalities managing *H. mantegazzianum* populations over time (excluding those that do not have populations in their boundaries)

3.4 Public perceptions of *H. mantegazzianum invasions and management*

Overall just under half (46%) of respondents knew the species by name – in French. Knowledge was generally greater amongst farmers with two (out of 11) not knowing *H. mantegazzianums* name. Knowledge of the species increased to 60% when picture was included. People who knew the plant generally categorised *H. mantegazzianum* as having a moderate abundance (57%), with others describing it as frequently occurring (32%) and rare (11%) in the study region. The majority of respondents who knew the plant said populations are stable (49%), or decreasing (18.7) with a number saying (34 %) populations were increasing in certain areas. This links quite closely to responses from municipalities and the trends found in the field sampling (section 3.2).

Regarding impact, most people viewed it as undesirable (97%) and said it had negative effects or posed a threat for them, specifically relating to human health impacts. Only one respondent mentioned it specifically having both benefits (ornamental/ aesthetic) and costs in their life.

In terms of policy and management, 72% of people who recognised the plant knew that is was on the black list. Only one of the 11 farmers noted the species is currently present on his land and did mention implementing control every year but viewed his overall management success rate to be low. Other respondents also mentioned having managed it and other IAS in the past on their property or in the vicinity of their residence. The majority of farmers (81%) mentioned they do control other IAS on their land annually and would likely do so for *H. mantegazzianum* if it spread onto their property. Overall, 81% of people who knew the plant supported eradication of the species from Switzerland. In general people who did not support control were against the use of chemical treatment, rather than the removal of the plant. By far the majority of all respondents preferred mechanical methods over other control approaches.

4. Discussion

Overall, large parts of the lower lying areas of Switzerland are climatically suitable for *H. mantegazzianum* to invade (Figure 1 and 2). In our smaller case study region of the Vaud pre-Alps our findings highlight that many parts of the region could also be highly susceptible to invasion, however, good management in most communes (but not all) is preventing this. Our results show that public and officials are well aware of the threat of this invasive plant and management is taking place in most areas (Figure 4) leading to reduced spread of *H. mantegazzianum* and even population declines or eradication in some areas– although in a few places populations are increasing. Our vegetation sampling corroborates this and highlights an overall decline in presence of *H. mantegazzianum* in most places, but spread at a few sites. This suggests that management has been successful, however, more needs to be done in some locations to control invasions and prevent spread to those areas that are well managed.

4.1 The need for predicting and monitoring invasive species population change and control efforts

Globally effective monitoring is a challenge with regards to IAS control and is often poorly implemented or not conducted at all (King and Downey 2008; Shackleton et al. 2016; van Wilgen et al. 2017; Latombe et al. 2018), which is also an issue is other environmental management programes globally (Turner et al. 2016). In this study, like others, we illustrate the value of engaging with local managers and the use of integrated methodological approaches to guide monitoring and assessing management progress (Blossey 1999; Manly et al. 2002; Conroy et al. 2008; Lee et al. 2008; Bryce et al. 2011; Peyrand et al. 2013). Monitoring of conservation, natural resource management and restoration projects is particularly important to inform and/or help to introduce adaptive management (Lyons et al. 2008), and it can also help with prioritisation (triage) of management approaches and areas (Downey et al. 2010).

In this study, we applied different methods to estimate the potential distribution and risk of spread of *H. mantegazzianum*, and monitored population changes on the ground over time. The predictive SDMs, fitted at different scales were informative of potential distribution, allowing anticipation of further spread and identifying high potential risk areas (Jiménez-Valverde et al. 2011; Gallien et al. 2012; Petitpierre et al. 2013) (Figure 1 and 2). The SDMs showed wide potential for invasion in Switzerland, and help to confirm common knowledge of important variables facilitating the distribution of the species, giving insights into factors that drive its presence and absence in a landscape, thus helping to understand its ecology. The latter model, fitted at a fine resolution, further allowed us to better conduct *H. mantegazzianum* population sampling and to monitor invasions in the field over time. Using such models can help to guide and target surveys which can save valuable time and money regarding population level monitoring and therefore increasing cost effectiveness (i.e., management recommendations could highlight that monitoring should be focused in area below 2000 m.a.s.l.).

The high potential for spread based on the SDM done in 2005 (later updated in 2013), led to a campaign of raising awareness, particularly aimed at commune officials in 2006. Questionnaire results with managers in 2013 and 2018 indicated that this has led to increased management efforts in most municipalities (Figure 4). Follow up plant population monitoring in 2013 and 2018 has shown great success with known established populations declining in most municipalities (especially lower-lying and more urban ones). Such successful management provides encouraging evidence that long-term control measures can be effective in reducing the size and cover of invasive plants (Nielsen et al. 2005). However, in a few places, unmanaged populations are growing considerably, as seen in areas within the municipalities of Ollon and Ormont-Dessus, requiring urgent management action, particularly as these populations are at higher altitudes and so can act as a seed source to areas that are well managed at lower altitudes.

The use of adaptive sampling methods for trying to estimate total population of *H. mantegazzianum* in the Vaud pre-Alps had mixed successes and we would advise some important considerations using these methods in the future. Six occurrences (out of 100 randomly stratified points) of *H. mantegazzianum* were found using adaptive sampling in 2005 and led to an estimated $\hat{u}_{st}$ x N = 3 300 000 ± 29 000 individuals in the study area, a substantially higher estimate than calculated in 2013 ($\hat{u}_{st}$ x N = 950 000), although only one large population was found and so no confidence intervals could be calculated. In 2018, we were unable to perform calculations as not a single population was found. This implies one of two things: 1) these estimates are imprecise and there are issues with the use of the approach – particularly in the case of small populations that are on average in decline. This illustrates that applying adaptive sampling methods can sometimes be challenging. In future use, adding stopping rules (a maximum number of plots per population cluster) could be considered to reduce bias if one of the sampling points occurs at an unusually large population (Brown 1994), as happened in the 2013 sampling period in our study. Additionally, issues can arise of not finding populations due to population declines (i.e. IAS being managed or declines through disturbance (Pergl et al. 2012)) and therefore the initial sample size should be increased or the search areas widened. 2) it provides further evidence of effective management of the plant in the study area and can be seen as a useful approach to illustrate management success. Despites the issues with this approach, evidence through the use of multiple different methods confirms that the number of invasive populations has decreased in the study area since 2005 due to an increased management effort by commune authorities post 2007 (Figure 4). Therefore, future work looking at rare and declining populations using adaptive sampling should also include other monitoring approaches as well to help confirm trends.

Similar declines in regional populations of *H. mantegazzianum* have also been observed in the Czech Republic but over a much larger time scale (Pergl et al. 2012). Of the total number of 521 historical sites at which the IAS has occurred since the end of the 19th century, it persists at only 124 (23.8%). The persistence rate differs with respect to habitat type and is highest in meadows and forest margins. Analysis using classification trees indicated that the factors that best explain persistence are: type of habitat (with meadow and forest margins having better persistence); urbanity (with a higher persistence outside urban areas); proximity to the place of the species' introduction into the country; metapopulation connectivity; and distance to the nearest neighbouring population. Pergl et al. (2012) attribute changes in *H. mantegazzianum* populations over time to potentially increased management in the past few decades, but also land use changes and urban expansion that have displaced invasive populations. In one case, in our study we saw that a field had been ploughed over an area that had previously been a site of invasion in Switzerland. We, however, attribute most declines over the last 13 years to increased management particularly after an awareness raising effort with municipalities in 2006. Active control using mowing, cutting and herbicides is evident and reducing population expansion and spread, and has led to some localised eradications of the plant in specific municipalities. *Heracleum mantegazzianum* is easier to eradicate locally than many other IAS, as it has lower persistence rates (Pyšek et al. 2001). These lower persistence rates are due to the fact that *H. mantegazzianu* is a short-lived monocarpic perennial which only reproduces by seed and often requires environmental distance or specific microclimates to establish (Pergl et al. 2012).

4.3 Future considerations and management of H. mantegazzianum in the Vaud-pre-Alps

For subregions or municipalities in the Vaud pre-Alps where management of *H. mantegazzianum* is proving effective and populations are in decline, it is recommended that follow-up management is maintained every year in the foreseeable future and monitoring for new invasions continues. This is needed due to the fact that *H. mantegazzianum* has high levels of seed production (an average plant producing ~20 000 seeds; Pergl et al. 2006; Perglová et al. 2006) and viability, and due to the fact seed remain dormant in the soil for up to seven years (Krinke et al. 2005; Moravcová et al. 2006; Moravcová et al. 2018). In areas where *H. mantegazzianum* is not being controlled or is poorly managed, the large spreading populations can act as a seed source to surrounding areas especially through dispursal along water courses and road networks, and pose a continued threat to human well-being and the environment (Thiele and Otte

2007). These large spreading populations need to be better managed by authorities, to reduce propagule pressure that can hamper effective control and progress elsewhere (Pergl et al. 2011). A particularly crucial consideration is that municipalities where management is not as efficient and invading populations' densities are higher, are commonly found at higher altitudes (compared to those in the Rhône Valley at the western edge of the study area). This means that *H. mantegazzianum* populations present there can easily disperse seed downstream along tributary rivers to lower located municipalities, where bioclimatic conditions are equally (or more) favourable. This can hamper management efforts in lower lying areas and so control efforts in these higher municipalities needs to be prioritised. Establishment of priority management areas should take into account such considerations, and focus on populations that are near to vectors for seed transport, such as riverbanks and roadsides (Pyšek and Prach 1994; von der Lippe and Kowarik 2007). In the future, methods such as multi-criteria decision making or decision trees could be used to prioritise populations for control in municipalities that do not have the capacity to address all populations and where there is need for triage (Downey et al. 2010; Forsyth et al. 2012; Shackleton et al. 2017). The results from the questionnaires also show the support for management by the public but that they generally do not actively control invasions themselves. Therefore, another management approach through engaging society to build awareness and collaborative control efforts between different stakeholders could help to better control the spread of remaining *H. mantegazzianum* populations and increase long-term buy-in towards management (Bryce et al. 2011; Novoa et al. 2018; Shackleton et al. 2019). Encouraging management of *H. mantegazzianum* by private land owners would help to reduce the burden on the state and may aid control efforts in higher altitude rural municipalities where capacity to manage invasions was noted as a major issue in questionnaire responses.

4.3 Increasing uptake of mixed methods approaches for research, monitoring and management of IAS globally

Lastly this study highlights the usefulness of integrating different social and ecological methods and approaches together, to improve holistic understanding of invasion dynamics and their management. Monitoring both plant population though ecological surveys and management implantation though social surveys at the same time helped to identify reasons for changing invasion population dynamics. As biological invasions are a coupled human-environmental phenomena we believe that use of integrated methods and interdisciplinary approaches should increasingly be applied within the field to facilitate improved understanding, which is currently lacking considerably (Vaz et al. 2017). There is scope for innovative research with useful practical implications from management in this area.

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