Environmental Impact Assessment Review 65 (2017) 1–11

Contents lists available at ScienceDirect

Environmental Impact Assessment Review

journal homepage: www.elsevier.com/locate/eiar

Environmental risk assessment for invasive alien species: A case study of apple snails affecting ecosystem services in Europe

Gianni Gilioli [a,*], Gritta Schrader [b], Nils Carlsson [c], Ellen van Donk [d], Casper H.A. van Leeuwen [e], Pablo R. Martín [f], Sara Pasquali [g], Montserrat Vilà [h], Sybren Vos [i]

[a] DMMT, University of Brescia, Viale Europa 11, 25123 Brescia, Italy
[b] Julius Kuehn Institute, Messeweg, 11-12, 38104 Braunschweig, Germany
[c] The County Administrative Board, Environmental Department, 20515 Malmoe, Sweden
[d] Netherlands Institute of Ecology (NIOO-KNAW), Droevendaalsesteeg 10, 6708 PB Wageningen, Netherlands
[e] Ecology and Biodiversity Group, Utrecht University, Netherlands
[f] INBIOSUR (Universidad Nacional del Sur-CONICET), San Juan 670, Bahía Blanca, Argentina
[g] CNR-IMATI, Via Bassini 15, 20133 Milano, Italy
[h] Estación Biológica de Doñana (EBD-CSIC), Avda. Américo Vespucio s/n, Isla de la Cartuja, 41092 Sevilla, Spain
[i] European Food Safety Authority, Via Carlo Magno, 1A, 43100 Parma, Italy

ARTICLE INFO

Article history:
Received 19 November 2016
Received in revised form 26 March 2017
Accepted 27 March 2017
Available online xxxx

Keywords:
Risk management
Pest risk analysis
Pomacea spp.
Plant pests
Population abundance
Wetlands

ABSTRACT

The assessment of the risk posed by invasive alien species (IAS) to the environment is a component of increasing importance for Pest Risk Analysis. Standardized and comprehensive procedures to assess their impacts on ecosystem services have been developed only recently. The invasive apple snails (*Pomacea canaliculata* and *P. maculata*) are used as a case study to demonstrate the application of an innovative procedure assessing the potential impact of these species on shallow freshwater ecosystems with aquatic macrophytes in Europe. The apple snail, *Pomacea maculata*, recently established in the Ebro delta in Spain resulting in a serious threat to rice production and wetlands, having also a high risk to spread to other European wetlands. Here, the population abundance of apple snails is regarded as the main driver of ecosystem change. The effects of ecosystem resistance, resilience and pest management on snail population abundance are estimated for the short (5 years) and the long (30 years) term. Expert judgment was used to evaluate the impacts on selected ecosystem services in a worst-case scenario. Our study shows that the combined effects of apple snails are estimated to have profound effects on the ecosystem services provided by shallow, macrophyte-dominated ecosystems in Europe. This case study illustrates that quantitative estimates of environmental impacts from different IAS are feasible and useful for decision-makers and invasive species managers that have to balance costs of control efforts against environmental and economic impacts of invasive species.

1. Introduction

Biological invasions are frequently considered processes of ecological disturbance (Turner, 2010) disrupting the structure of the community, the population dynamics, and changing the resource availability or the physical environment (Pickett and White, 1985). Disturbances alter the state of an ecosystem and its trajectory; they are key drivers of spatial and temporal heterogeneity (Turner, 2010). However, the effects of a well-established invasive alien species (IAS) in a new territory cannot solely be considered as a disturbance; due to the temporal persistence of the IAS in the receiving environment their presence also represents a driver of ecosystem change. IAS are recognized among the five most important direct or structural drivers of ecosystem change (Henrichs et al., 2010; Tomich et al., 2010) and can affect the provision of ecosystem services significantly.

Ecosystems provide important services to humans. Ecosystems services (ES) are for example the provisioning of freshwater, food production and genetic resources. IAS often disrupt or alter ES. To what extent this might occur depends on the particular species and the ecosystem. Therefore, it remains challenging to estimate the potential impact of IAS on ES.

To make accurate management decisions regarding control of IAS, it is necessary to assess their potential impact both for the short and the long term. In this paper we demonstrate a novel procedure for the

* Corresponding author.
E-mail addresses: gianni.gilioli@unibs.it (G. Gilioli), gritta.schrader@julius-kuehn.de (G. Schrader), E.vandonk@nioo.knaw.nl (E. van Donk), c.v.leeuwen@ibv.uio.no (C.H.A. van Leeuwen), sara@mi.imati.cnr.it (S. Pasquali), montse.vila@ebd.csic.es (M. Vilà), Sybren.VOS@efsa.europa.eu (S. Vos).

http://dx.doi.org/10.1016/j.eiar.2017.03.008
0195-9255/

assessment of environmental impacts caused by IAS to provide required information supporting decision making by the risk managers. The method provides a comprehensive and integrated environmental risk assessment (ERA) that is applicable to any kind of IAS by combining the autecology of the species with the known functions of the invaded ecosystem.

The method was developed according to the guidance document on the ERA of the European Food Safety Authority (EFSA) in which the framework for an ERA was described in detail (EFSA, 2011). The method was further developed and applied in Gilioli et al. (2014). Compared to EFSA (2011) and Gilioli et al. (2014), the approach proposed here represents a substantial improvement towards the applicability of a fully quantitative approach to ERA based on ES. There are two main novelties. First, the causal chain linking the IAS to the impact on ES has been further clarified assigning the role of driving force modifying the ecosystem traits involved in the ES to the population abundance of the IAS and its spatial and temporal variability (Fig. 1). Second, an expert knowledge elicitation (EKE) procedure (EFSA, 2014b) has been successfully applied to the estimation of the probability distributions of the parameters influencing the population pressure and the impact for the selected relevant ES.

The aim of this paper is to illustrate this novel, standardized approach by a case study, assessing the potential impact of apple snails' populations on a selected group of ES that are provided by shallow freshwater ecosystems with aquatic macrophytes.

Apple snails of the genus *Pomacea* have recently established and spread in Spain. Therefore, Europe was chosen as risk assessment area for this study. The method itself is applicable to other regions– it is only necessary to define the area for which the risk is assessed and to have available the climatic conditions and the relevant habitats for this area.

These snails are in the list of the 100 worst IAS in the world (see Global Invasive Species Database, 2015) and are known to devastate shallow freshwater ecosystems which provide important ES (Carlsson, 2006; Morrison and Hay, 2011). Therefore, they represent an ideal case study for testing the ERA framework. We concentrate our assessment on the island apple snail (previously identified as *Pomacea insularum* (d'Orbigny, 1835) and now described as *P. maculata*), and the channelled apple snail *P. canaliculata* (hereinafter referred to as *Pomacea* spp. or the apple snail (s)). These two species are closely related, have often been confounded and their population dynamics pattern and potential impacts are similar (Hayes et al., 2012; Horgan et al., 2014). These two species of apple snails are highly invasive outside their native distribution range in South America. They are serious rice pests (Joshi and Sebastian, 2006) and can have detrimental effects on the flora and fauna of natural freshwater wetlands by causing drastic declines in aquatic macrophytes (Carlsson et al., 2004). This is due to their characteristics, such as diverse diet, high feeding and reproduction rate, and the presence of specific adaptations like their capacity to breathe with both lungs and gills. Furthermore, they can survive adverse conditions by retreating into their shell and closing it firmly with the operculum (Horgan et al., 2014). By this, predators are discouraged and the snails can hibernate or aestivate buried in mud within the protective moisture of their shell for periods of up to eleven months when their habitat dries out (Oya et al., 1987; Yusa et al., 2006).

The apple snail *P. maculata* has been accidentally released in the Ebro delta in Spain (Anon., 2011).There it has established viable populations and is spreading, leading to significant damage to crops (Anon., 2011, EFSA, 2012) and representing a serious threat for ES and biodiversity. After its first outbreak reported in 2009, the snail, that was not known to occur in Europe before, continues its invasion in the Ebro delta and now also in Toll del Vidre, Arnes (Tarragona), despite mechanical and chemical control measures, inundation of rice paddies with saline water and other methods to eradicate or contain it in the rice paddies (Anon., 2011, EFSA, 2012). Currently, the snail is present not only in rice paddies, but also in some nearby wetlands, and it has been found moving upwards along the Ebro river. The import and trade of apple snails were banned by the European Union in 2012, but *P. canaliculata*, and probably also *P. maculata*, are nevertheless still sold online for the aquarium trade within the risk assessment area (Mazza et al., 2015). Hence, there is great concern that the snail might establish in other parts of Europe.

The aim of this paper is to provide a detailed estimation of the potential impact of the apple snail populations on a selected group of ES that

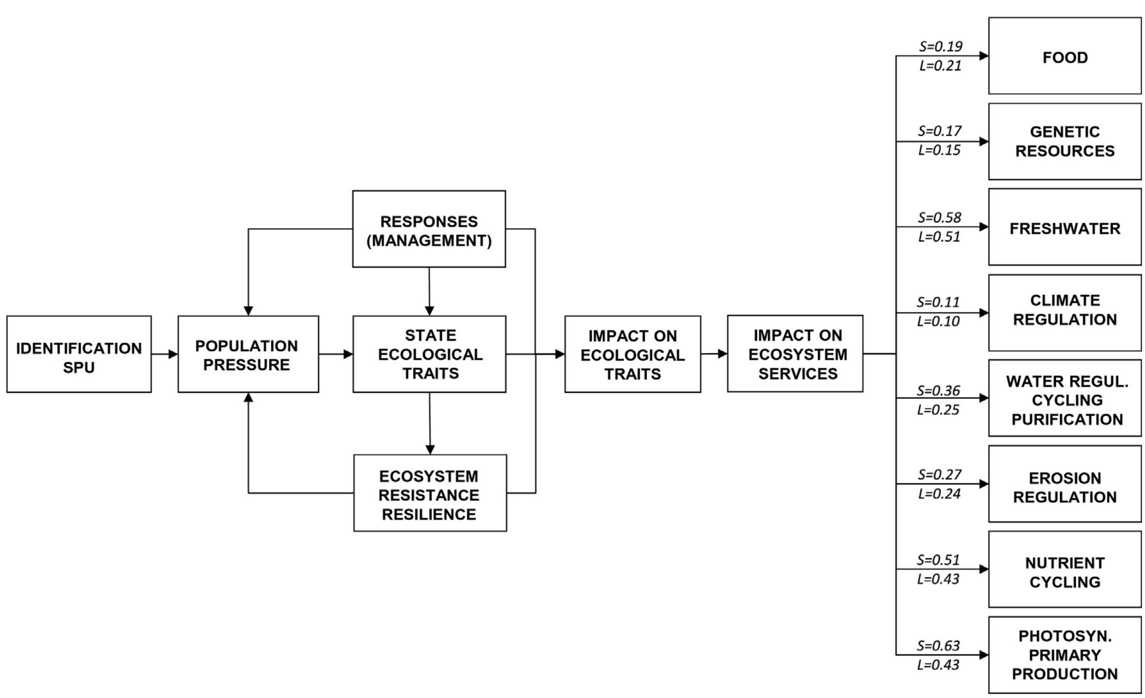

Fig. 1. Causal chain linking the changes in the Service Providing Unit to the impact on Ecosystem Services due to population pressure of the Invasive Alien Species. The risk posed by *Pomacea* spp. to the selected Ecosystem Services in Europe, as described in Section 3.2, in the short term (S) and in the long term (L), is reported numerically beside each box representing an ecosystem service.

are provided by shallow freshwater ecosystems with aquatic macrophytes. The assessment takes into account previously published information on the snail biology and ecology of the two species, as well as the environmental impact that the snails have caused elsewhere. Wetland ecologists participated in an expert knowledge elicitation procedure to allow the estimation of the impacts on ES. The approach also considers the uncertainties regarding the estimations of the impact. The assessment was performed considering the worst case scenario for the risk assessment area (defined as the maximum population abundance potentially attained in Europe, that is 31.5 g/m^2), and for the short and the long term period (defined by the experts as 5 and 30 years after establishment, respectively).

2. Methodology

The ERA presented in this paper is built on a scenario analysis (Gilioli et al. 2014). The change in the ES provision level is assessed by collecting the experts' judgment. The following provisioning ES were considered: food, genetic resources, and freshwater (as water quality/quantity for human consumption). As regulating and supporting ES, climate regulation, water regulation/cycling/purification, erosion regulation, nutrient cycling, photosynthesis and primary production of macrophytes were assessed (MEA, 2003).

2.1. Scenario assumptions

2.1.1. Identification of the service providing unit

The impact of the snail is related to the environmental components or units responsible for the genesis and regulation of the ES, the so-called service-providing units (SPU, see Luck et al., 2003). The structural and functional characteristics of the SPU represent the state of the system before the invasion and allow defining the constraints and possibilities of ecosystem change after invasion.

For the definition of the SPU, the homogeneity of the type of ES being provided by the ecosystem is important, irrespective of the homogeneity of the environment containing the aquatic plants susceptible to attack by the apple snail (mostly submersed and floating macrophytes). This broad approach will focus and speed up environmental risk assessment of invasive species substantially regardless the identity of the invader. Apple snails are essentially freshwater animals and their reproductive habits (aerial egg-laying) and their dual respiration system determine their preference for shallow and shoreline areas (Seuffert and Martín, 2010, 2013). Here, a single SPU is considered to be affected by *Pomacea* spp., i.e. shallow freshwater ecosystems, which include wetlands, shallow lakes, river deltas and the littoral zone of deeper lakes and rivers. Although shallow freshwater ecosystems can be ecologically diverse, they share a homogeneous environment in such a way that macrophytes offer retention and processing of nutrients and toxic substances, physical structure, habitat, refuge, food or substrate for food and a substrate for spawning of invertebrates, fish and amphibians (van Donk and van De Bund, 2002). Although rice fields are also important for the provision of ES (Natuhara, 2013), these are not included in the SPU. The interaction between the SPU and the cultivated areas however is taken into account in this assessment, regarding, for example, the exchange of water and the movement of organisms.

2.1.2. Population pressure

The snail population abundance, expressed in terms of biomass per area unit, in the area of potential establishment represents the population pressure modifies the ecosystem structure and traits with effect on the ES' level of provision.

A physiologically based demographic model had been developed for the species *P. canaliculata*, since there is more bio-ecological information available for this species (EFSA, 2013). The similarities in the biology of *P. canaliculata* and *P. maculata* allow considering the population abundance predicted by this model as an estimate of population pressure of the two apple snails. The model describes the area of potential establishment of the apple snails in Europe and predicts the distribution of their abundance as a function of spatial variation in the environmental forcing variables. The availability of information on the population abundance allows defining the areas in Europe where the snail biomass (population pressure) reaches the highest levels. The maximum biomass is estimated for the south-west of Spain where the potential abundance reaches 31.5 g/m^2 of wet weight of juveniles and adults.

2.1.3. Scales of the analysis

The scenario analysis requires the definition of the spatial and the temporal scale at which the assessment is performed. The definition of the spatial and temporal scales depends on the objectives of the assessment. The state and the dynamics of the recipient ecosystem are also considered in the choice of these scales. Here, we have not defined a spatial scale but have considered only the worst-case scenario for the analysis. This scenario considers an ideal service-providing unit (SPU) representing the 'average condition' of macrophyte dominated freshwater habitats in Europe that are most favourable for the apple snail (i.e., where the snail biomass per area unit reaches the maximum level or potential abundance, and where the highest environmental impact is supposed to occur according to the information available from already infested areas (e.g., Carlsson et al., 2004; Joshi and Sebastian, 2006). For the temporal scale, two time horizons have been selected: short term (5 years after establishment) and long term (30 years after establishment).

2.1.4. State and reactions of the receiving ecosystem

The assessment of the environmental consequences of *Pomacea* spp. induced transformation of shallow freshwater, macrophyte-dominated ecosystems in Europe must address the interaction between the invader and the receiving communities and ecosystems. The impact of the apple snail can be modified or mitigated by the changes in the community and the ecosystem functioning of the recipient ecosystem. The apple snail population pressure may change over time depending on three factors which are a) the ecosystem's resistance, i.e. the capability of the ecosystem to remain relatively functionally intact despite the disturbance from *Pomacea* herbivory, b) the ecosystem's resilience, i.e. the capability of an ecosystem to return to its original state, as well as c) the pest management. For the scenario analysis specific assumptions (see below and Section 2.2.2) were made on the effects of these three factors.

We derived the realized biomass from the potential abundance estimated by means of the population dynamics model. The latter is computed multiplying the value of the potential biomass by coefficients defined here as scaling factors. These scaling factors (ranging from 1, no effect, to 0, maximum effect) take into account the effects of resistance, resilience and management as a reaction of the receiving ecosystem or to the implementation pest control measures which consequently reduce the potential biomass to the realized biomass. The values of the scaling factors are estimated by means of an expert judgment elicitation procedure as described below.

The effects of resistance, resilience and management over time require the consideration of different scenarios with respect to the temporal scale. Two temporal horizons have been taken into account. In 5 years after establishment, the population dynamics of the snail should reach their maximum level in the most favourable conditions in Europe and are mainly influenced by the resistance of the receiving environment, and by the containment and eradication efforts. Thirty years after establishment the resilience e.g. occurrence or dominance of apple snail-resistant macrophytes (Morrison and Hay, 2011; Wong et al., 2010) is supposed to play a more important role, as well control by predators and other natural enemies which need to adapt to the presence of the exotic apple snail (Yamanishi et al., 2012; Yusa et al., 2006). More specific management measures are required and thus considered. General changes in climate (e.g., water temperatures and frequency of droughts) and wetland habitats that are not related to the invasion have not been considered in the 30-year assessment scenario.

2.2. The collection of expert judgments

The assessment of the impacts of apple snails on ES was done by a panel of five experts with knowledge of the biology and ecology of the apple snails and of the ecology of macrophyte dominated shallow freshwater ecosystems. They were consulted to obtain information on the impact on the ES subject to assessment and on the scaling factors. The experts were supported by guidelines presenting the information necessary to perform the assessment based on the EFSA Guidance on Expert Knowledge Elicitation in Food and Feed Safety Risk Assessment (EFSA, 2014b). Two rounds of consultation were performed. In the first round the experts individually provided probability distribution for each of the scaling factors and ES to be assessed, and supported their estimations by explanations and references. Results of the first round were made available to all the experts. In the second round, starting from the results obtain in the first turn of consultation a common discussion allowed to obtain an agreed distribution for each of the assessed scaling factors and ES.

In the following subsections we present the rating system adopted in this study to perform the assessment (see also EFSA, 2014a). The rating system follows the scheme suggested in the PLH ERA guidance (EFSA, 2011).

2.2.1. Impact on ecosystem services

The impact on (ES) was evaluated through a discrete probability distribution estimating the percentage of reduction in the level of provision of the assessed ES, for both the short and the long term. Five different categories of impact (minimal, minor, moderate, major and massive) have been considered. Each category of impact is defined by an interval of percentage of reduction of the ES as indicated in Table 1.

For each ES the experts had to provide the probability distribution of the reduction in ES provision level, as indicated in Table 1. The values p_1, p_2, p_3, p_4, p_5 (representing the probability of the corresponding impact) must be non-negative and such that $p_1 + p_2 + p_3 + p_4 + p_5 = 1$.

As representative point for each interval of percentage reduction, the mid-point of the interval has been chosen (Table 1) and used to calculate the risk for an ES. The risk associated with ES *i* was calculated as follows (EFSA, 2011).

$$R_i = 0.025\, p_2 + 0.125\, p_3 + 0.35\, p_4 + 0.75\, p_5.$$

Finally, the risk can be categorised, and five classes have been considered (Table 1).

A common distribution of the impact for each ES starting from expert judgments has been derived by the following procedure. The reduction in each ES provision was evaluated by K experts. Let denote by $\{p_1^k, p_2^k, p_3^k, p_4^k, p_5^k\}$ the probability distribution of the k^{th} expert. The probability distributions given by all the experts were combined using a mixture distribution (see Johnson et al., 1992, p. 53) that weights the evaluation of the different experts. A final distribution was obtained (Table 2), where w_k are weights satisfying the following properties.

$$w_k > 0 \;\; k = 1, 2, \ldots, K \;\; and \;\; \sum_{k=1}^{K} w_k = 1.$$

In the present study, the expert judgments were equally weighted, then $w_k = 1/K$.

The final risk was calculated on the mixture distribution that takes into account all the experts evaluations.

Experts' evaluations are subject to uncertainty. To assess the uncertainty associated with the evaluation of the impact on an ES the Shannon entropy (Shannon, 1948) has been used as proposed in EFSA (2011).

$$U_i = -\sum_{j=1}^{5} p_j \log\left(p_j\right).$$

Variable U_i was normalised with respect to its maximum, that is $\log(J)$.

$$U_i^* = \frac{U_i}{\log(J)}.$$

The uncertainties U_i^* can be categorised, and three classes have been considered (Table 3).

2.2.2. Scaling factors

For each of the scaling factors (resistance, resilience and management) the experts were asked to provide a 95% confidence interval of the mean value previously fixed on the basis of discussion between experts. In other words, they had to indicate an interval in which the scaling factor falls with 95% probability.

The experts' evaluations were then combined to obtain a single 95% confidence interval for the mean of each scaling factor. The scaling factor is a variable ranging between 0 and 1. A typical distribution to model the behaviour of variables taking values in the interval [0,1] is the beta distribution (Johnson et al., 1992, p. 210). The estimated mean and 95% confidence interval allows obtaining the parameters of the beta distribution for the corresponding scaling factor. This can be done for all the experts. All the beta distributions obtained for a fixed scaling factor were combined in a mixture distribution. From this final distribution a single 95% confidence interval has been obtained for the scaling factor that summarizes all the experts' evaluations.

Table 1
Ratings of reduction in ecosystem services provision level (i.e., Impact) and representative points (defined as mid-points) for each interval; probability assigned to each of the five ratings, and categories for the risk.

	Rating				
	Minimal	Minor	Moderate	Major	Massive
Impact	Zero or negligible	]0%, 5%]	]5%, 20%]	]20%, 50%]	]50%, 100%]
Representative point	0	0.025	0.125	0.35	0.75
Probability	p_1	p_2	p_3	p_4	p_5
Risk	Zero or negligible	]0, 0.05]	]0.05, 0.20]	]0.20, 0.50]	]0.50, 1]

3. Results and discussion

3.1. Scaling factors

3.1.1. Resistance

Herbivory by *Pomacea* spp. has been shown to induce a shift from clear water and macrophyte dominance towards turbid waters, increased nutrient concentrations and phytoplankton dominance (Carlsson et al., 2004). These effects of the snail invasion are similar to the already ongoing effects of eutrophication in many European waters. Due to large-scale eutrophication over the last 200 years in Europe, many shallow freshwater, macrophyte-dominated ecosystems in Europe have become less resistant to further disturbance (de Nie, 1987; Sand-Jensen et al., 2000). In addition, European macrophytes lack a co-evolutionary history with *Pomacea* spp., and should therefore have a higher susceptibility to snail herbivory than macrophytes in the native range of *Pomacea* spp. (Morrison and Hay, 2011). On the short term, it is unlikely that natural enemies (in particular avian species breeding in the Ebro delta, for instance the glossy ibis (*Plegadis falcinellus*) and the yellow-legged gull (*Larus michahellis*)) will be able prevent the snail's population establishment and growth. For these reasons, ecosystem

Table 2
Probability distribution of the k^{th} expert ($k = 1,2,...K$), and mixture distribution obtained combining the evaluations of the different experts.

	Rating				
	Minimal	Minor	Moderate	Major	Massive
Reduction	Zero or negligible	]0%, 5%]	]5%, 20%]	]20%, 50%]	]50%, 100%]
Probability	p_1^k	p_2^k	p_3^k	p_4^k	p_5^k
Mixture	$\sum_{k=1}^{K} w_k p_1^k$	$\sum_{k=1}^{K} w_k p_2^k$	$\sum_{k=1}^{K} w_k p_3^k$	$\sum_{k=1}^{K} w_k p_4^k$	$\sum_{k=1}^{K} w_k p_5^k$

resistance to a *Pomacea* spp. invasion is expected to be very low in Europe for both the short and long term. The estimated values of the resistance parameter are 0.9 for the short term and 1.0 for the long term. This means that we expect a population abundance of 90% of the maximum expected abundance after 5 years and of 100% of the maximum abundance after 30 years (Table 4).

3.1.2. Resilience

The resilience of the freshwater ecosystems may increase over time. On the short term, natural enemies are not expected to play an important role, but in the long term several natural enemy species may start to use and even specialise on *Pomacea* spp. as a new abundantly available food source (Carlsson et al., 2009). On longer terms, non-palatable macrophytes may become dominant due to differential herbivory by the snails (Horgan et al., 2014; Meza-Lopez and Siemann, 2015). We expect reductions in many fish and bird species as a response to the decline in macrophytes in the systems, which would further reduce the important predators of *Pomacea* spp. as a secondary result of *Pomacea* spp. herbivory. The estimated values of the resilience parameter are 0.95 for the short term and 0.5 for the long term. This means that we expect a population abundance of snails of 95% of the maximum expected abundance after 5 years and, due to effects of resilience, of 50% of the maximum abundance after 30 years (Table 4).

3.1.3. Management

Management has the potential to lower the apple snail population abundance and spread but some management methods may also cause an additional impact on the environment. Risk reduction options that are applied to rice fields may cause negative environmental effects on the adjacent and connected natural systems (assessed here as the SPU), for example:

(1) Keeping rice paddies dry for a long period (Wada, 2004). This might negatively influence rice paddy biodiversity, in particular soil biodiversity and birds visiting the rice ecosystem.
(2) Burning vegetation and removal of plants along river banks of rice fields to prevent egg laying and survival of snails. This will have a negative effect on flora and fauna of river ecosystems when applied on a large scale and over several years.
(3) Treating rice paddies with saponins, lime and saline water. This could result in negative effects on both the rice and the natural SPU ecosystems (Anon., 2011, EFSA, 2012), however, these effects have found to be transient by Joshi et al. (2008).

Because of the proximity of the rice paddies and the natural ecosystems all the risk reduction options mentioned under 1–3 above seem to result in serious negative effects on the SPU and, therefore, should not be used. Some other methods to control apple snails in rice paddies might also be used in the SPU, e.g. hand or mechanical collection of snails and installation of snail traps, though it is as yet unknown how snail traps affect other biota. The negative effects of control measures aimed at apple snails are expected to be only a fraction of the negative environmental effects caused by other management methods used to control pests, diseases and weeds in rice production areas. Available information on the control of rice pests suggests that the current methods used to control the apple snail might result in only minor additional negative effects (EFSA, 2012; Joshi et al., 2008). The biotic resistance towards the apple snail by local predators or competitors may be enhanced by preserving the natural complexity of microhabitats or by introducing modifications that reduce the availability of refuges for the apple snail. As investigated by Hara et al. (2015), the semi-natural and natural canals have a lower probability of containing waterweed than the artificial canals. The various microhabitats of the natural canals are known to maintain a greater number of animal species such as the removal of aquatic plants like waterweed that functions as a shelter for apple snails.

Based on the information presented above and on expert estimates, the effect of management measures on reduction of potential snail biomass in freshwater wetlands is estimated to be low in the short term and moderate in the long term. The estimated values of the management parameter are 0.99 for the short term and 0.8 for the long term. This means that we expect a population abundance of 99% of the maximum expected abundance after 5 years and of 80% of the maximum abundance after 30 years (Table 4).

3.1.4. Combination of the scaling factors

The scaling factors have been estimated in the two time horizons. The product of the coefficients for resilience, resistance and management assumes the value of 0.84 for the short term and 0.40 for the long term (Table 4). This means that we expect a population abundance of snails of 84% of the maximum expected abundance of 31.5 g/m^2 after 5 years (corresponding to 26.5 g/m^2) and, due to the effects of the scaling factors, of 40% of the maximum population abundance after 30 years (corresponding to 12.6 g/m^2).

Table 3
Categories for the uncertainty.

	Rating		
	Low	Medium	High
Uncertainty	]0, 0.33]	]0.33, 0.67]	]0.67, 1]

Table 4
Mean values and confidence intervals (CI) of the scaling factors for the ecosystem's resistance and resilience and the pest management as well the population abundance of the snails for the short and the long term used in the assessment of the snails' impacts on ecosystem services.

	Short term: 5 years	Long term: 30 years
Mean scaling factors		
Resistance (*RS*)	0.90 (CI [0.8242;0.9758][a])	1.00
Resilience (*RL*)	0.95 (CI [0.9021;0.9979][a])	0.50 (CI [0.3593;0.6407][a])
Management (*MN*)	0.99 (CI [0.9563;1][a])	0.80 (CI [0.6607;0.9393][a])
$RS \times RL \times MN$	0.84	0.40
Abundance		
Maximum potential abundance	31.5 g/m^2	31.5 g/m^2
Maximum realized abundance	26.5 g/m^2 (normalised to the maximum = 0.84)	12.6 g/m^2 (normalised to the maximum = 0.40)

[a] 95% Confidence interval for the mean of the scaling factor.

3.2. Impact on ecosystem services

3.2.1. Food

The expected effects of increased snail biomass on food production (i.e. less amounts of food being available) are rated to be moderate to major (Fig. 2), both in the short term and in the long term. The reduction is predicted to increase somewhat in the long term even though we expect a lower realized snail abundance due to increased resilience since many fish and bird species are long-lived and the long-term effects of lowered production of offspring should therefore be more pronounced than the effects on individuals.

The European wetlands are often highly productive and some of this natural production is used for human consumption. Fish and birds are mainly used for consumption, but also other organisms, for example freshwater crayfish, may also be important food locally. The functions and qualities of these habitats are further crucial to other food production such as livestock production (easily accessible drinking water of good quality) and aquaculture in both freshwater and coastal areas.

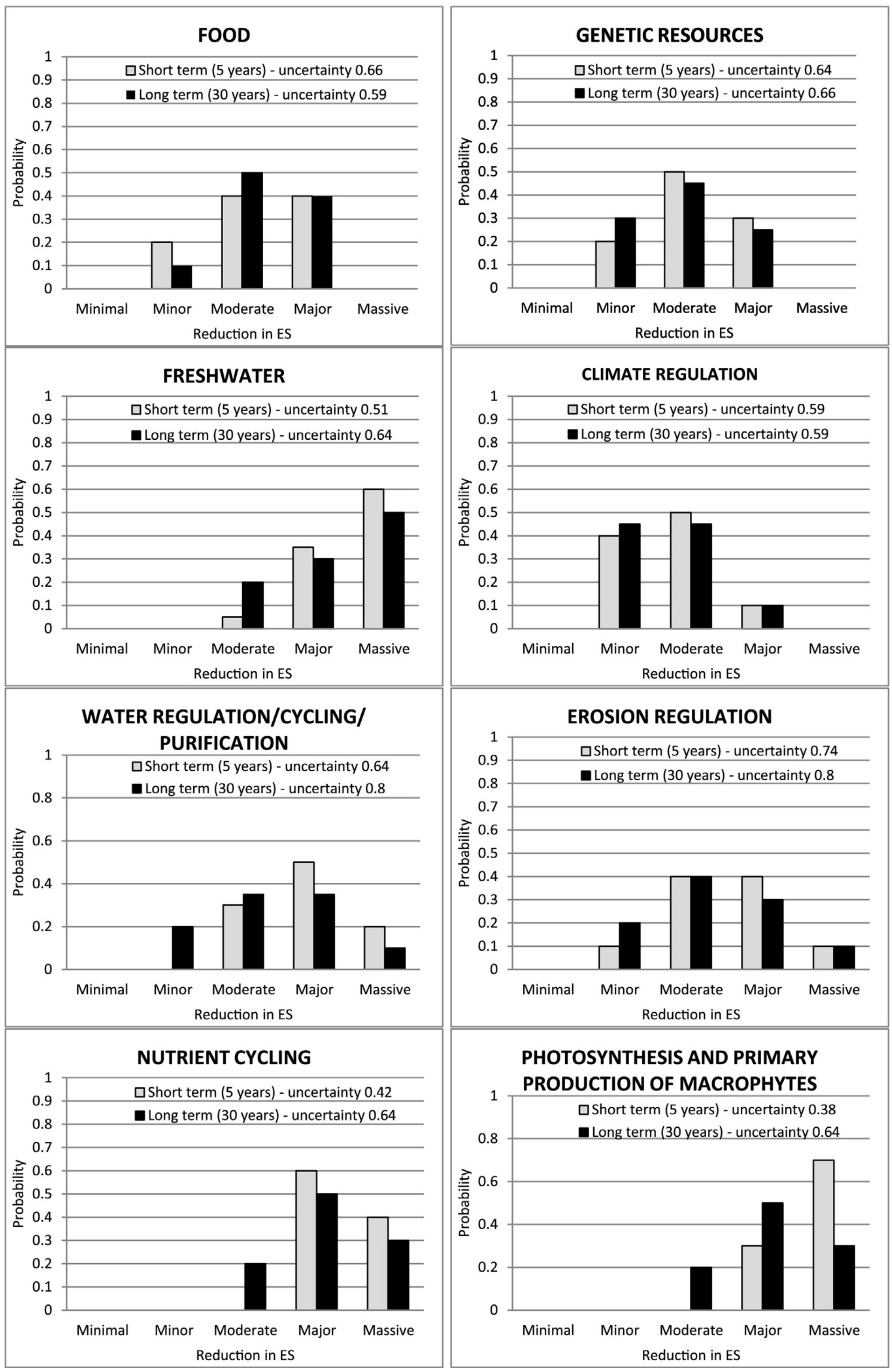

Fig. 2. Probability distribution (obtained as mixture distribution of experts evaluations) and associated uncertainty of the reduction in the provision level of the ecosystem services. The experte estimates were performed based on the worst-case scenario (maximum realized snails biomass) in the short and long term.

Pomacea spp. herbivory is expected to affect several ecosystem traits that in turn affect food production. Fish, bird and crayfish production is directly affected since many fish, bird and crayfish species include macrophytes as an important component in their diet, or consume the invertebrates that thrive in the macrophytes (Hansson et al., 2010; Kaenel et al., 1998).

Many fish species use the vegetated shallows seasonally for reproduction and as nurseries for juvenile fish, and most waterfowl species forage in these areas. The abundance and diversity of macrophytes affect fish production in several ways, including reduced availability of the preferred spawning substrate and thus reduced production of offspring (Wychera et al., 1993), less shelter from predators and lower survival rates when the fish larvae emerge from the eggs (Zohary and Ostrovsky, 2011). These negative effects on several fish and bird species may be very pronounced. Loss of foraging areas negatively affects bird recruitment at a larger scale. It has been demonstrated in long term studies that several waterfowl species adjust their migratory routes and actively avoid lakes with low macrophyte coverage (Hansson et al., 2010). Following the introduction of invasive and voraciously herbivorous crayfish (*Procambarus clarkii*), in a lake in Spain, the sharp reductions in macrophytes lead to rapid and secondary declines in phytophagous birds and 75% losses in duck species (Rodríguez et al., 2005). *P. clarkii* removes all macrophytes and has disrupting effects on freshwater ecosystems. However, this species also eats snails (Klose and Cooper, 2012), including apple snails, so herbivorous crayfish to some extend control apple snail biomass, but at the cost of having largely similar effects on freshwater ecosystems.

The highest impact on food production, however, is if herbivory by apple snails induces an ecosystem shift from clear water and macrophyte dominance towards turbid waters and increased nutrient concentrations and phytoplankton dominance. It has previously been demonstrated that reductions in macrophyte abundance and macrophyte species diversity that have resulted from eutrophication have induced large shifts in European fish communities (Wolter et al., 2000). Furthermore, in waters that receive high nutrient loads, herbivory enhanced eutrophication may result in extensive fish kills at the end of the summer growing season from the deoxygenation that arises from the decomposition of dead algae (e.g. Søndergaard et al., 1999). Increased deposition of dead algae on spawning sites may also reduce littoral fish populations by suffocating incubating eggs. In the worst case scenario, the systems get dominated by blooming, toxic phytoplankton that are directly harmful for birds, mammals and fish and consequently, all food production.

There are several uncertainties to take into consideration when estimating the negative effects of *Pomacea* spp. on food production. The expected shifts in fish towards less predatory, phytophilic fish species (Wolter et al., 2000) and bird communities towards less waterfowl species that are herbivores, invertebrate, or fish feeders (Hansson et al., 2010) may be compensated partially by increases in other fish or bird species. It is, for example, likely that cyprinid fish species and omnivorous waterfowl will increase (Hansson et al., 2010; Wolter et al., 2000). It is uncertain, however, to which extent this would happen. From a food production perspective such “compensation” may still be problematic since cyprinids generally are of less commercial value than piscivorous fish. In addition, a dominance of cyprinids accentuates eutrophication due to their promotion of algal production both by their consumption of zooplankton and by their resuspension of the nutrients in the sediments (Meijer, 2000).

3.2.2. Genetic resources

Genetic resources include the genes and genetic information used for animal and plant breeding and biotechnology (MEA, 2003). Increasing biomass of the apple snail is predicted to have a major to massive effect on genetic resources in the short term, and a major effect on genetic resources in the long term (Fig. 2). We forecast two main processes that will generate reductions in genetic resources with increasing abundance of *Pomacea* spp.; the first is through reductions in, or losses of, aquatic species populations, the second, through a reduced connectivity (increased isolation) between these shrinking populations. A drastic reduction in, or local or regional losses of, several macrophyte species populations, of course, constitutes a direct loss of genetic diversity and a loss of local or regional genetic adaptations. These reductions in macrophytes then induce secondary reductions in genetic diversity in the populations (or local or regional losses) of several groups of organisms that depend on macrophytes at any life stage. Increasingly smaller and more isolated populations of several species inevitably lead to loss in low frequency alleles, lowering the species' ability to adapt to changing environments (Jump et al., 2009). In recent years, the importance of bird-mediated dispersal of aquatic organisms in gene transport between aquatic populations has been increasingly acknowledged (Amezaga et al., 2002; Lurz et al., 2002; van Leeuwen et al., 2012; van Leeuwen et al., 2013). However, since many species of waterfowl avoid waterbodies with low macrophyte coverage (Hansson et al., 2004), herbivory by apple snails may further weaken connectivity between aquatic populations and further impacts regional genetic diversity negatively (Amezaga et al., 2002).

The uncertainties for these predictions are medium and relate in particular to difficulties in predicting the future development and presence of species ability to live in habitats with a reduced abundance of macrophytes. An increase in non-palatable plants and an increase in predation of apple snails could increase bird mediated dispersal of organisms and lower the impact in the long term.

3.2.3. Freshwater

The effects of *Pomacea* on water quantity in wetlands, rivers and lakes are not expected to be very important. However, the retention time of water at larger geographic scales is expected to decrease slightly if macrophytes decline, because beds of aquatic plants physically impede water movement and increase the water holding capacity of the landscape.

The reduction in water quality, though, is expected to be massive both in the short term and in the long term (Fig. 2), especially in shallow, macrophyte dominated systems. Species rich and abundant macrophyte communities' play a key role in nutrient cycling, act as important natural biofilters and ensure a base level of water quality even at high nutrient loads (Carlsson, 2006; Petr, 2000). The most serious consequence on water quality occurs when the aquatic ecosystems that already receive high nutrient loads are pushed towards a turbid water state with dominance of planktonic algae. When the macrophytes are reduced drastically, it is no longer possible to remove nutrients or toxic heavy metals from the system by harvesting macrophytes. Current efforts to decrease nutrient inputs to freshwater systems through improved wastewater treatment and changed farming practices will also be less effective in achieving clear water and (returning) macrophyte dominance if the re-emerging macrophytes again are consumed by *Pomacea* spp. as has been suggested for North American wetlands by Burlakova et al. (2009). Likewise, efforts to restore macrophyte communities in North-western Europe are now jeopardized when the emerging macrophytes are rapidly consumed by the invasive crayfish *Procambarus clarkii* (van Der Wal et al., 2013).

The uncertainty of these ratings is in the medium range and relates to the presence of other factors than macrophytes that also influence water quality such as the magnitude and type of nutrient load, water depth, sediment composition and several other factors. The impact of *Pomacea* spp. on water quality, should, for example, be greater in wetlands, compared to larger, deeper lakes where macrophytes only are present in a small fraction of the total water volume.

3.2.4. Climate regulation

The effects of *Pomacea* spp. on climate regulation in the short and long term are concentrated in the moderate/minor range (Fig. 2). Wetlands and lakes are both sinks (through carbon assimilation and retention) and sources (through methane (CH_4) and nitrogen dioxide (N_2O) release) of gases that affect climate regulation. Freshwater snails have been shown to promote methane release from wetlands (Xu et al., 2014). A recent meta-analysis shows that emissions of both the greenhouse gases CH_4 and N_2O is significantly lower in areas with abundant macrophytes compared to open water areas. Increasing biomass of *Pomacea* spp. will instead reduce macrophytes, and therefore decrease carbon assimilation and retention and increase emissions of the greenhouse gases CH_4 and N_2O.

The uncertainty of these ratings is in the medium range and relates to difficulties in predicting the possible long-term increase in non-palatable macrophytes and the uncertainties about the relative importance of macrophytes for climate regulation in deeper lakes compared to the well-known role played by macrophytes in wetlands.

3.2.5. Water regulation/cycling/purification

The effects of *Pomacea* on water runoff, flooding and aquifer recharge are expected to be less important than the effects on the ecosystem's capacity to filter and purify chemical waste, and organic pollution since macrophytes are vital in all these processes, as mentioned earlier. The anticipated effects of an increasing apple snail abundance on water purification are described in detail in the previous Section 3.2.3 “Freshwater”. In the short term we expect a major reduction in these purification processes. In the long term, a less dramatic reduction (moderate to major) is predicted since an increase in non-palatable macrophytes may offer some compensation for the loss of palatable macrophytes (Fig. 2).

The uncertainty for the short term is medium but it is high for the long term since a shift towards phytoplankton dominance hampers an increase also in the non-palatable macrophytes. This scenario does not allow the purification processes to be restored. Increased predation on *Pomacea* spp. by native predators may regulate the invasive population at levels where plant recovery is possible.

3.2.6. Erosion regulation

The reduction in erosion regulation as a result of *Pomacea* spp. is expected to be between moderate and major in the short term and slightly more moderate than major in the long term (Fig. 2) since an increase in non-palatable macrophytes may offer some compensation for the loss of root area in palatable macrophytes. Macrophytes have a strong effect on erosion in fluvial water systems since they stabilize the sediments with their roots and impede water velocity, and therefore, increase sedimentation (Madsen et al. 2001). Macrophytes are further light-limited and grow predominantly in the shallow littoral zone close to the shore or river bank in deeper systems. If they are consumed, we expect not only increased erosion of river sediments but also increased erosion of the river bank itself. Both sources of erosion will increase nutrient and sediment loads transported to downstream systems and finally, coastal areas. This effect should be less pronounced in lakes and wetlands than in rivers, but drastic declines in macrophytes should increase both wind and wave erosion at the shoreline of lakes and wetlands when the water velocity and waves are no longer impeded by beds of macrophytes.

The uncertainty of these predictions is high and relates to the earlier mentioned increased resilience and rebound of macrophytes.

3.2.7. Nutrient cycling

The effects of *Pomacea* spp. on nutrient cycling are predicted to be major to massive in the short term (Fig. 2). A shift from macrophyte dominance to phytoplankton dominance reduces nutrient burial in the sediments, may induce anoxia at the sediment surface, with subsequent release of phosphorus, and increase resuspension of sediments. As mentioned earlier, this internal load of phosphorous may be far greater than the external load of phosphorous (Istanovics et al., 2004). This may cause a release of large quantities of phosphorus and nitrogen to downstream aquatic ecosystems, causing increased eutrophication and coastal hypoxia.

Macrophytes assimilate nitrogen, phosphorus and heavy metals directly, impede water movement and increase sedimentation of particle-bound substances, which allows nutrients and harmful substances to be processed, or buried permanently, in the sediment. They further reduce nitrogen levels by providing a huge substrate for periphytic algae that host denitrifying bacteria that transform nitrate to N_2 gas, which leaves the water. They further remove phosphorus by facilitating processes such as sorption, precipitation, direct uptake and peat/soil accretion (Vymazal, 2007). Their nutrient cycling and shading of sunlight also hampers phytoplankton growth and, subsequently, prevents the development of harmful and extremely costly toxic algal blooms (Pretty et al., 2003). If the water flow is less impeded by macrophytes, sedimentation is decreased and the sediments become less stable and more easily re-suspended by water and wind movements (Koch, 2001). Declines in macrophytes typically lead to subsequent increases in phytoplankton concentrations (Scheffer et al., 1993) and decaying phytoplankton consumes large amounts of oxygen. When the sediment surface becomes anoxic, phosphorus is no longer chemically bound to the sediment and instead released to the water column, promoting new algal blooms, this internal load of phosphorous may be far greater than the external load of phosphorous (Istanovics et al., 2004). This is a major concern since eutrophication, as mentioned earlier, of many freshwater systems in Europe is an ongoing process due to increased aerial deposition of nutrients, waste water input and run-off from fertilised farmlands.

Uncertainty in the short term is medium. In the long term we still expect a major effect with medium uncertainty on nutrient cycling but again an increase of non-palatable macrophytes could offer some compensation for the loss of palatable macrophytes.

3.2.8. Photosynthesis and primary production of macrophytes

The reduction in primary production and photosynthesis by macrophytes due to *Pomacea* is predicted to be massive in the short term and major in the long term (since some non-palatable macrophyte species may increase), both rated with medium uncertainty (Fig. 2).

3.3. Overall impact on ecosystem services

Table 5 summarizes the risk and uncertainty for the selected ecosystem services. The risk of invasive apple snails for genetic resources and climate regulation is moderate in both the short and the long terms. The risk for food production is moderate in the short term, but it becomes major in the long term. The risk for water regulation and erosion regulation is major in both the short and the long terms. The risk for nutrient cycling, photosynthesis and primary production of macrophytes is massive in the shorter, but declines to major in the long term. Finally, the risk for freshwater is massive in both the short and the long terms.

4. Conclusions

Our study illustrates the use of a risk assessment method based on an improved version of the method proposed in Gilioli et al. 2014 to estimate the potential impact of an invasive population on a selected group of ES. Depending on the scope and the conditions of the assessment, one or more SPU can be assessed, and the assessment can be tailored to the needs of the risk managers, thus, this method can be used to produce focused and time-efficient risk assessments' of IAS with known ecology and biology by identifying a SPU that provide ES that will be effected by the invader.

By considering a (in this case single) In this case study, a single SPU was considered, for which we merged wetlands, shallow lakes, river deltas and the littoral zone of deeper lakes and rivers, to estimate environmental impacts of an invading species for a broad range of freshwater ecosystems.

Merging the different habitats to one SPU was possible since aquatic macrophytes are the key elements in freshwater ecosystem structure

Table 5
Risk and uncertainty for ecosystem services.

	Short term				Long term			
	Risk		Uncertainty		Risk		Uncertainty	
Ecosystem service	Value	Category	Value	Category	Value	Category	Value	Category
Food	0.19	Moderate	0.66	Medium	0.21	Major	0.59	Medium
Genetic resources	0.17	Moderate	0.64	Medium	0.15	Moderate	0.66	Medium
Freshwater	0.58	Massive	0.51	Medium	0.51	Massive	0.64	Medium
Climate regulation	0.11	Moderate	0.59	Medium	0.10	Moderate	0.59	Medium
Water regulation/cycling/purification	0.36	Major	0.64	Medium	0.25	Major	0.80	High
Erosion regulation	0.27	Major	0.74	High	0.24	Major	0.80	High
Nutrient cycling	0.51	Massive	0.42	Medium	0.43	Major	0.64	Medium
Photosynthesis and primary production of macrophytes	0.63	Massive	0.38	Medium	0.43	Major	0.64	Medium
Overall	0.35	Major	0.57	Medium	0.29	Major	0.67	Medium

and functioning (Jeppesen et al., 1998a, 1998b). Carlsson et al. (2004) have previously demonstrated that the invasion by *Pomacea canaliculata* in Asian wetlands dramatically reduces both the species richness and the abundance of macrophytes. Apart from apple snails' invasion, these reductions are also being observed as a general trend in freshwater habitats in Europe and may generate fundamental reductions (or even losses) in the structure and functions of freshwater habitats in Europe, and subsequently, reductions (or even losses) in the ecosystem services these habitats provide. At high *Pomacea* spp. abundance the risk of an ecosystem shift from clear water and macrophyte dominance towards turbid waters and phytoplankton dominance is of major concern since such ecosystem shifts are not easily reversible (Scheffer et al., 1993). The recolonization potential of macrophytes in turbid, phytoplankton dominated waters is very low and remaining apple snails, which are able to persist using quite diverse trophic resources other that macrophytes (Saveanu and Martín, 2015), will make this undesirable shift even less reversible by consuming the few macrophytes that do recolonize.

Our case study clearly shows that an unhalted spread of *Pomacea* spp. in Europe has a drastic potential impact on several important ES that are provided by a wide range of freshwater habitats in Europe. Although the risks for genetic resources and climate regulation are estimated to be moderate, we expect a massive impact on freshwater quality, and major impacts on erosion regulation, nutrient cycling and primary production of macrophytes, as well as on food production in the long term. In the worst case scenario, the overall effect of the snail invasion on the shallow freshwater ecosystems of southern Europe is major on the ecosystem services both in the short and in the long terms (see Table 1).

Our predictions of risks for ecosystem services include specific uncertainties for each ecosystem service. There is, however, a general uncertainty that underlies all predictions, namely, whether there are macrophyte species non-palatable to the apple snail in the assessment area (i.e. those species with high contents of phenolic compounds and relatively low contents of nutrients (low C/N ratio) (Qiu et al., 2011; Wong et al., 2010). If so, these may increase when palatable species decrease and offer some compensation for the loss of palatable macrophytes and of the structures and functions that these macrophytes used to provide. Although such a shift in macrophyte dominance is difficult to forecast, there are several reasons to believe that this compensation will be insufficient.

European macrophytes lack a co-evolutionary history with *Pomacea* spp. and previous studies suggest that this may give a high susceptibility to snail herbivory. Apple snails prefer naive North American macrophytes that have not been previously exposed to *Pomacea* spp. herbivory to South American macrophytes that have coevolved with apple snails, since the North American macrophytes lack the chemical and physical defenses that deter the snails (Morrison and Hay, 2011). A shift towards phytoplankton dominance may further hamper an increase in the non-palatable macrophytes as these become light-limited. It has been demonstrated that some of the less palatable plants are more easily consumed by *Pomacea* spp. when in a decaying state (Qiu et al., 2011), which could accelerate the increase in apple snail populations and the eradication of macrophytes communities. On the other hand, predation on apple snails by native predators may become an important factor in the long term that regulates the snail populations at a level where recovery of macrophytes is possible. For all the above reasons, we expect compensation by non-palatable plants to be low and difficult to forecast. This adds a general uncertainty to our predictions.

In this paper the proposed method has been applied to the worst case scenario, which corresponds to a specific SPU with optimal biotic and abiotic environmental conditions allowing the population to reach the maximum abundance. Despite the uncertainty and variability related to the complexity that underlie ecosystem processes, the approach we propose allows the assessment of impacts of an IAS at ecosystem level for all suitable habitats in a large risk assessment area, i.e. Europe, and at different level of population abundance. The method allows producing maps of the impact based on of the expected distribution of the population abundance of the IAS as provided by a population dynamic s model, and considering the distribution of the suitable habitats, each characterized by site-specific environmental conditions. The availability of functions relating the impact of the IAS to the population abundance can support the translation of the spatial pattern of population abundance into spatial pattern of environmental risk.

The approach identifies areas where research would be necessary to increase knowledge on the species and its impacts. In some cases there is a lack of knowledge in the relationships linking population abundance and the ecosystem traits related to the provision of ES. The definition of the clusters interpreting the functional relationship between ecosystem traits and services is often the most difficult passage in the application of the assessment scheme we propose. Long term studies on invasion impact are also important in understanding the role of ecosystem resilience, and in setting the scaling factors the approach requires to estimate the temporal pattern of the environmental risk. In our case study, for example, more knowledge is needed on how palatable various macrophytes are to apple snails to better estimate which macrophytes species might be resilient to the herbivory and may remain in infested freshwater areas. Furthermore, knowledge is needed on the different dispersal modes of *Pomacea* spp., the possibility for rapid co-evolution between the snails and macrophytes, and the response of predators to the presence of the snails.

Quantitative estimates of environmental impacts are urgently needed for decision makers and invasive species managers that are facing decisions where they have to weigh the cost of large scale control efforts of an invader against its environmental and economic impact. The proposed framework allows generating comparable and reproducible results and enables the assessment of overall environmental risk caused by any kind of IAS. The procedure based on expert knowledge elicitation provides a cost-efficient synthesis of the available data and information and produces scenario-based projection of the impact at the selected level of spatial and temporal resolution. The proposed assessment scheme is also a powerful tool in selecting potential IAS for the

European black list that is requested by the European invasive species directive (EU, 2014).

Acknowledgements

The authors acknowledge the EFSA Panel on Plant Health and the whole staff of the EFSA PLH Team for encouraging the development and application of ecosystem-based approaches for the environmental risk assessment of plant pests. The positions and opinions presented in this article are those of the authors alone and are not intended to represent the views or scientific works of EFSA.

References

Amezaga, J.M., Santamaría, L., Green, A.J., 2002. Biotic wetland connectivity supporting a new approach for wetland policy. Acta Oecol. 23 (3), 213–222.

Anon., 2011. Pest Risk Analysis on the Introduction of *Pomacea insularum* (d'Orbigny, 1835) Into the EU. Prepared by the Spanish Ministry of Environment and Rural and Marine Affairs (April).

Burlakova, L.E., Karatayev, A.Y., Padilla, D.K., Cartwright, L.D., Hollas, D.N., 2009. Wetland restoration and invasive species: apple snail (*Pomacea insularum*) feeding on native and invasive aquatic plants. Restor. Ecol. 17 (3), 433–440.

Carlsson, N.O.L., 2006. Invasive Golden Apple Snails are Threatening Natural Ecosystems in Southeast Asia. In: Joshi, R.C., Sebastian, L.S. (Eds.), Global Advances in Ecology and Management of Golden Apple Snails. Philippine Rice Research Institute, Muñoz, Philippines, Nueva Ecija, pp. 61–72.

Carlsson, N.O.L., Brönmark, C., Hansson, L.A., 2004. Invading herbivory: the golden apple snail alters ecosystem functioning in Asian wetlands. Ecology 85, 1575–1580.

Carlsson, N.O.L., Sarnelle, O., Strayer, D.L., 2009. Native predators and exotic prey – an acquired taste? Front. Ecol. Environ. 7 (10), 525–532.

de Nie, H.W., 1987. The decrease in aquatic vegetation in Europe and its consequences for fish populations. Rome Italy: EIFAC/CECPI Occasional Paper No. 19.

EFSA (European Food Safety Authority), 2014b. Guidance on expert knowledge elicitation in food and feed safety risk assessment. The EFSA Journal 12 (6), 3734 (278 pp.).

EFSA (European Food Safety Authority) Panel on Plant Health (PLH), 2011. Guidance on the environmental risk assessment of plant pests. The EFSA Journal 9 (12), 2460 (121 pp.).

EFSA (European Food Safety Authority) Panel on Plant Health (PLH), 2013. Scientific opinion assessment of the potential establishment of the apple snail in the EU. The EFSA Journal 11 (12), 3487 (50 pp.).

EFSA (European Food Safety Authority) Panel on Plant Health (PLH), 2014a. Scientific opinion on the environmental risk assessment of the apple snail for the EU. The EFSA Journal 12 (4), 3641 (97 pp.).

EFSA Panel on Plant Health (PLH), 2012. Scientific opinion on the evaluation of the pest risk analysis on *Pomacea insularum*, the island apple snail, prepared by the Spanish Ministry of Environment and Rural and Marine Affairs. The EFSA Journal 10 (1), 2552 (57 pp.).

EU, 2014. Regulation (EU) no. 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species. Official Journal of the European Union L 317 (57), 35–55.

Gilioli, G., Schrader, G., Baker, R., Ceglarska, E., Kertész, V., Lövei, G., Navajas, M., Rossi, V., Tramontini, S., van Lenteren, J., 2014. Environmental risk assessment for plant pests: a procedure to evaluate their impacts on ecosystem services. Sci. Total Environ. 468-469, 475–486.

Global Invasive Species Database, 2015. Pomacea canaliculata.Available at. http://www.iucngisd.org/gisd/species.php?sc=135 (Accessed 18 September 2016).

Hansson, L.A., Gyllström, M., Stahl-Delbanco, A., Svensson, M., 2004. Responses to fish predation and nutrients by plankton at different levels of taxonomic resolution. Freshw. Biol. 49, 1538–1550.

Hansson, L.A., Nicolle, A., Brönmark, C., Hargeby, A., Lindström, Å., Andersson, G., 2010. Waterfowl, macrophytes, and the clear water state of shallow lakes. Hydrobiologia 646 (1), 101–109.

Hara, A., Hamasaki, K., Yoshida, K., Yusa, Y., 2015. Canal type affects invasiveness of the apple snail *Pomacea canaliculata* through its effects on animal species richness and waterweed invasion. Biol. Invasions 17 (1), 63–71.

Hayes, K.A., Cowie, R.H., Thiengo, S.C., Strong, E.E., 2012. Comparing apples with apples: clarifying the identities of two highly invasive Neotropical Ampullariidae (Caenogastropoda). Zool. J. Linnean Soc. 166 (4), 723–753.

Henrichs, T., Zurek, M., Eickhout, B., Kok, K., Raudsepp-Hearne, C., Ribeiro, T., Van Vuuren, D., Volkery, A., 2010. Scenario development and analysis for forward-looking ecosystem assessments. In: Ash, N., Blanco, H., Brown, C., Garcia, K., Henrichs, T., Lucas, N., Raudsepp-Hearne, C., David Simpson, R., Scholes, R., Tomich, T., Vira, B., Zurek, M. (Eds.), Ecosystems And Human Well-Being: A Manual for Assessment Practitioners. Island Press, London, UK.

Horgan, F.G., Stuart, A.M., Kudavidanage, E.P., 2014. Impact of invasive apple snails on the functioning and services of natural and managed wetlands. Acta Oecol. 54, 90–100.

Istvanovics, V., Osztoics, A., Honti, M., 2004. Dynamics and ecological significance of daily internal load of phosphorus in shallow Lake Balaton, Hungary. Freshw. Biol. 49 (3), 232–252.

Jeppesen, E., Sondergaard, M., Jensen, J.P., Mortensen, E., Hansen, A.M., Jorgensen, T., 1998a. Cascading trophic interactions from fish to bacteria and nutrients after reduced sewage loading: an 18-year study of a shallow hypertrophic lake. Ecosystems 1, 250–267.

Jeppesen, E., Søndergaard, M., Sondergaard, M., Christofferson, K. (Eds.), 1998b. The Structuring Role of Submerged Macrophytes in Lakes. 131. Springer Science & Business Media.

Johnson, N.L., Kotz, S., Balakrishnan, N., 1992. Continuous Univariate Distributions. 1–2. John Wiley and Sons, New York.

Joshi, R.C., San Martín, R., Saez-Navarrete, C., Alarcon, J., Sainz, J., Antolin, M.M., Sebastian, L.S., 2008. Efficacy of quinoa (*Chenopodium quinoa*) saponins against golden apple snail (*Pomacea canaliculata*) in the Philippines under laboratory conditions. Crop. Prot. 27 (3), 553–557.

Joshi, R.C., Sebastian, L.S., 2006. Global Advances in Ecology and Management of Golden Apple Snails. Philippine Rice Research Institute, Nueva Ecija.

Jump, A.S., Marchant, R., Penuelas, J., 2009. Environmental change and the option value of genetic diversity. Trends Plant Sci. 14 (1), 51–58.

Kaenel, B.R., Christoph, D., Matthaei, C.D., Uehlinger, U.R.S., 1998. Disturbance by aquatic plant management in streams: effects on benthic invertebrates. Regul. Rivers Res. Manag. 14 (4), 341–356.

Klose, K., Cooper, S.D., 2012. Contrasting effects of an invasive crayfish (*Procambarus clarkii*) on two temperate stream communities. Freshw. Biol. 57, 526–540.

Koch, E.W., 2001. Beyond light: physical, geological, and geochemical parameters as possible submersed aquatic vegetation habitat requirements. Estuaries 24, 1–17.

Luck, G.W., Daily, G.C., Ehrlich, P.R., 2003. Population diversity and ecosystem services. Trends Ecol. Evol. 18, 331–336.

Lurz, P.W.W., Shirley, M.D.F., Rushton, S.P., Sanderson, R.A., 2002. Modelling the consequences of duck migration patterns on the genetic diversity of aquatic organisms: a first step towards a predictive tool for wetland management. Acta Oecol. 23 (3), 205–212.

Madsen, J.D., Chambers, P.A., James, W.F., Koch, E.W., Westlake, D.F., 2001. The interaction between water movement, sediment dynamics and submersed macrophytes. Hydrobiologia 444, 71–84.

Mazza, G., Aquiloni, L., Inghilesi, A.F., et al., 2015. Aliens just a click away: the online aquarium trade in Italy. Manag. Biol. Invasions 6 (3), 253–261.

MEA (Millennium Ecosystem Assessment), 2003. Ecosystems and human well-being: a framework for assessment. Report of the Conceptual Framework Working Group of the Millennium Ecosystem Assessment. Island Press, Washington, DC.

Meijer, M.-L., 2000. Biomanipulation in the Netherlands: 15 Years of Experience. Wageningen Universiteit, The Netherlands.

Meza-Lopez, M.M., Siemann, E., 2015. Experimental test of the invasional meltdown hypothesis: an exotic herbivore facilitates an exotic plant, but the plant does not reciprocally facilitate the herbivore. Freshw. Biol. 60 (7), 1475–1482.

Morrison, W.E., Hay, M.E., 2011. Herbivore preference for native vs. exotic plants: generalist herbivores from multiple continents prefer exotic plants that are evolutionarily naïve. PLoS One 6 (3), e17227.

Natuhara, Y., 2013. Ecosystem services by paddy fields as substitutes of natural wetlands in Japan. Ecol. Eng. 56, 97–106.

Oya, S., Hirai, Y., Miyahara, Y., 1987. Overwintering of the apple snail, *Pomacea canaliculata* Lamarck, in north Kyushu, Japan. Jpn. J. Appò. Entomol. Z 31, 206–212.

Petr, T., 2000. Interactions between fish and aquatic macrophytes in inland waters: a review. Rome: Food and Agriculture Organisation (FAO). FAO Fisheries Technical Reports, Paper 396 (185 pp.).

Pickett, S.T., White, P.S. (Eds.), 1985. The Ecology of Natural Disturbance and Patch Dynamics. Academic Press, Orlando, FL.

Pretty, J.N., Mason, C.F., Nedwell, D.B., Hine, R.E., Leaf, S., Dils, R., 2003. Environmental costs of freshwater eutrophication in England and Wales. Environ. Sci. Technol. 37, 201–208.

Qiu, J.W., Chan, M.T., Kwong, K.L., Sun, J., 2011. Consumption, survival and growth in the invasive freshwater snail *Pomacea canaliculata*: does food freshness matter? J. Molluscan Stud. 77 (2), 189–195.

Rodríguez, C.F., Bécares, E., Fernández-Aláez, M., Fernández-Aláez, C., 2005. Loss of diversity and degradation of wetlands as a result of introducing exotic crayfish. Biol. Invasions 7 (1), 75–85.

Sand-Jensen, K., Riis, T., Vestergaard, O., Larsen, S.E., 2000. Macrophyte decline in Danish lakes and streams over the past 100 years. J. Ecol. 88, 1030–1040.

Saveanu, L., Martín, P.R., 2015. Neuston: a relevant trophic resource for apple snails? Limnologica 52, 75–82.

Scheffer, M., Hosper, S.H., Meijer, M.-L., Moss, B., Jeppesen, E., 1993. Alternative equilibria in shallow lakes. Trends Ecol. Evol. 8, 275–279.

Seuffert, M.E., Martín, P.R., 2010. Dependence on aerial respiration and its influence on microdistribution in the invasive freshwater snail *Pomacea canaliculata* (Caenogastropoda, Ampullariidae). Biol. Invasions 12 (6), 1695–1708.

Seuffert, M.E., Martín, P.R., 2013. Distribution of the apple snail *Pomacea canaliculata* in Pampean streams (Argentina) at different spatial scales. Limnologica 43 (2), 91–99.

Shannon, C.E., 1948. A Mathematical Theory of Communication. Reprinted with corrections from. Bell Syst. Tech. J. 27 pp. 379–423, 623–656.

Søndergaard, M., Jensen, J.P., Jeppesen, E., 1999. Internal phosphorus loading in shallow Danish lakes. Hydrobiologia 408 (0), 145–152.

Tomich, T.P., Argumedo, A., Baste, I., Camac, E., Filer, C., Garcia, K., Garbach, K., Geist, H., Izac, A.M., Lebel, L., Lee, M., Nishi, M., Olsson, L., Raudsepp-Hearne, C., Rawlins, M., Scholes, R., van Noordwijk, M., 2010. Conceptual frameworks for ecosystem assessment: their development, ownership, and use. In: Ash, N, Blanco, H, Brown, C, Garcia, K, Henrichs, T, Lucas, N, Raudsepp-Hearne, C, David Simpson, R, Scholes, R, Tomich, T, Vira, B, Zurek, M (Eds.), Ecosystem and human well-being: a manual for assessment practitioners. Island Press, Washington DC, USA, pp. 71–113.

Turner, M.G., 2010. Disturbance and landscape dynamics in a changing world. Ecology 91 (10), 2833–2849.

van der Wal, J.E., Dorenbosch, M., Immers, A.K., Forteza, C.V., Geurts, J.J., Peeters, E.T., Bakker, E.S., 2013. Invasive crayfish threaten the development of submerged macrophytes in lake restoration. PLoS One 8 (10), e78579.
van Donk, E., van De Bund, W., 2002. Impact of submerged macrophytes including charophytes on phyto- and zooplankton communities: allelopathy versus other mechanisms. Aquat. Bot. 72 (3–4), 261–274.
van Leeuwen, C.H.A., Huig, N., van Der Velde, G., van Alen, T.A., Wagemaker, C.A.M., Sherman, C.D.H., Klaassen, M., Figuerola, J., 2013. How did this snail get here? Several dispersal vectors inferred for an aquatic invasive species. Freshw. Biol. 58, 88–99.
van Leeuwen, C.H.A., van Der Velde, G., van Groenendael, J.M., Klaassen, M., 2012. Gut travellers: internal dispersal of aquatic organisms by waterfowl. J. Biogeogr. 39, 2031–2040.
Vymazal, J., 2007. Removal of nutrients in various types of constructed wetlands. Sci. Total Environ. 380, 48–65.
Wada, T., 2004. Strategies for controlling the apple snail *Pomacea canaliculata* (Lamarck) (Gastropoda: Ampullariidae) in Japanese direct-sown paddy fields. JARQ-Jpn Agr Res Q 38 (2), 75–80.
Wolter, C., Minow, J., Vilcinskas, A., Grosch, U., 2000. Long-term effects of human influence on fish community structure and fisheries in Berlin waters: an urban water system. Fish. Manag. Ecol. 7, 97–104.
Wong, P.K., Liang, Y., Liu, N.Y., QIU, J.W., 2010. Palatability of macrophytes to the invasive freshwater snail *Pomacea canaliculata*: differential effects of multiple plant traits. Freshw. Biol. 55 (10), 2023–2031.
Wychera, U., Zoufal, R., Christof-Dirry, P., Janauer, G.A., 1993. Structure and environmental factors in macrophyte stands. J. Aquat. Plant Manag. 31, 118–122.
Xu, X., Zhao, W., Xiao, M., Huang, J., Fang, C., Li, B., Nie, M., 2014. Snails promote methane release from a freshwater lake ecosystem. Front. Environ. Sci. 2, 12.
Yamanishi, Y., Yoshida, K., Fujimori, N., Yusa, Y., 2012. Predator-driven biotic resistance and propagule pressure regulate the invasive apple snail *Pomacea canaliculata* in Japan. Biol. Invasions 14 (7), 1343–1352.
Yusa, Y., Wada, T., Takahashi, S., 2006. Effects of dormant duration, body size, self-burial and water condition on the long-term survival of the apple snail, *Pomacea canaliculata* (Gastropoda: Ampullariidae). Appl. Entomol. Zool. 41 (4), 627–632.
Zohary, T., Ostrovsky, I., 2011. Ecological impacts of excessive water level fluctuations in stratified freshwater lakes. Inland Waters 1, 47–59.