Environmental risk assessment for plant pests: A procedure to evaluate their impacts on ecosystem services


HIGHLIGHTS

- A new scheme to assess the impacts of plant pests and invasive alien species is presented.
- Impact is evaluated by estimating how plant pests modify ecosystem services.
- The assessment is based on a scenario analysis with defined spatial and temporal resolution.
- The proposed scheme is tested by using the Citrus long-horned beetle as an example.

ABSTRACT

The current methods to assess the environmental impacts of plant pests differ in their approaches and there is a lack of the standardized procedures necessary to provide accurate and consistent results, demonstrating the complexity of developing a commonly accepted scheme for this purpose. By including both the structural and functional components of the environment threatened by invasive alien species (IAS), in particular plant pests, we propose an environmental risk assessment scheme that addresses this complexity. Structural components are investigated by evaluating the impacts of the plant pest on genetic, species and landscape diversity. Functional components are evaluated by estimating how plant pests modify ecosystem services in order to determine the extent to which an IAS changes the functional traits that influence ecosystem services. A scenario study at a defined spatial and temporal resolution is then used to explore how an IAS, as an exogenous driving force, may trigger modifications in the target environment. The method presented here provides a standardized approach to generate comparable and reproducible results for environmental risk assessment as a component of Pest Risk Analysis. The method enables the assessment of overall environmental risk which integrates the impacts on different components of the environment and their probabilities of occurrence. The application of the proposed scheme is illustrated by evaluating the environmental impacts of the invasive citrus long-horned beetle, A. chinensis.

1. Introduction

The worldwide introduction and spread of invasive alien species (IAS) have led to important ecological consequences and are considered to be one of the major threats to biodiversity and ecosystem functioning (MA, 2005; Pascal et al., 2010). IAS have economic, environmental, social and health impacts, but often interest is limited to the economic and health impacts of IAS, while less attention is attributed to their environmental impacts (Parker et al., 1999).

Despite the importance of functional aspects involved in the action of IAS as a cause of ecosystem change, most analyses of environmental impact give greater importance to i) the consideration of the impact on components of the structural biodiversity, to account for the non-utilitarian value of nature, and ii) conservation related issues (Callicott et al., 1999). However, in the context of environmental risk assessment...
(ERA) of IAS, it is also crucial to consider the functionalist perspective (Callicott et al., 1999). This perspective focuses on the contribution of functional biodiversity in defining how systems cope with IAS as drivers of ecosystem change, and how IAS can drive ecosystem functions (and services) to a less desirable state (Walker, 1992; Naeem, 1998; Fonseca and Ganade, 2001; Rosenfeld, 2002).

The risks associated with plant pests are assessed in a structured procedure, known as Pest Risk Analysis (PRA) as described in the International Standards on Phytosanitary Measures (ISPMs), particularly in ISPM 11 (FAO, 2004). From the PRA point of view, the evaluation of environmental impacts is based on the analysis of scenarios related to on-going invasions or discrete entry, establishment and spread events. This requires a set of assumptions and knowledge of the possible future states and events in the ecosystem, including in particular the population dynamics of the plant pest and the effect on the species in the assessed area. A PRA also requires the assessment of overall environmental risk which integrates the impacts on different components and levels of the environment, and their probabilities of occurrence. In this case, overall risk cannot be expressed in classical ecological units and variables such as density, fluxes, and diversity. Therefore, the use of methods for measuring the overall environmental risk and the use of integrative variables (as a common ‘currency’) are required (Niemstedt et al., 2012).

ISPM 11 (FAO, 2004) provides only brief guidelines on how to assess the environmental impacts of plant pests by listing some examples of direct and indirect environmental consequences. However, the methods that are appropriate for environmental risk assessments are not indicated. Though the EPPO PRA scheme (EPPO, 2011), through the revision described by Kenis et al. (2012), now provides a set of detailed questions with explanatory notes for undertaking a qualitative assessment of environmental impacts, it does not provide an assessment based on ecosystem services since it is considered that many of these are directly related to social and economic impacts that are covered by other questions in the EPPO PRA scheme.

To approach environmental risk assessment in a well formulated theoretical context as well as providing practical guidance, we designed a scheme that assesses the functional components of the environment that might be threatened by plant pests. A risk assessment can only produce consistent, reproducible and comparable results when founded on standardized methods that are directly applicable. Recently, considerable progress has been made in the application of the ecosystem services concept to assess environmental impacts in many fields, and it is important to determine whether they can be applied effectively in plant biosecurity. We believe that an ecosystem service approach provides a suitable basis for the analysis of environmental impacts caused by IAS, because it accounts for both the impacts on components of structural biodiversity as well as ecosystem function. An ecosystem service approach also allows the impacts on the different components of the affected ecosystems to be combined by associating them with the probabilities of occurrence to obtain an overall index that summarizes environmental risk.

In this paper we explore a new method for evaluating the environmental consequences of the entry, establishment and spread of a plant pest. The method is not limited to plant pests (ISPM 5, FAO, 2012), but can be used for all types of invasive organisms. The definition of “plant pest” is related to the CBD definition (CBD, 2002) of an IAS, with the main difference that a plant pest is restricted to direct or indirect injuries to plants (FAO, 1998; Schrader and Unger, 2003), and the impact of a plant pest is not limited to impacts on biodiversity. In this paper, we use both terms a) because of their close relation, and b) since we consider the method is not limited to plant pests. With this method, both the risks posed to the structural and to the functional aspects of ecosystems can be assessed. A method for assessing the effects of IAS on structural biodiversity has recently been proposed by Kenis et al. (2012). Here we concentrate on a method for assessing impacts on the functional aspects of ecosystems. We present a conceptual framework that combines a description of environmental impact based on ecological processes with a scientific interpretation of the interface between natural ecosystem processes and human interactions. The proposed scheme has been tested by using a plant pest Anoplophora chinensis (Coleoptera: Cerambycidae) as an example and the results are briefly summarized.

2. Methodology

The procedure for an ERA of a pest is based on a chain of processes linking an invasion with the consequences for structural biodiversity and ecosystem services (Fig. 1). We first present the components in this chain of events and the processes, and then discuss the characteristics and properties of the mechanisms involved.

2.1. Plant pests as a driving force of ecosystem change

An invasion by a plant pest may trigger ecological disturbances that can cause potentially large changes in the affected ecosystems. If a new pest establishes, it can change its role into an ecological driving factor (ecological driver) and result in important modifications of the ecosystem. When an ERA is undertaken, it needs to take into account the way in which humans manage and influence the environment, including the effects of pest management. Thus, the consequences of pest invasions can be regarded as the effects of the pest on human-modified ecosystems (Western, 2001) or social–ecological systems (Walker and Salt, 2006). In the ERA for plant pests it is very important to consider environmental structures and processes, taking into account their interaction as a systemic complex unit. Therefore, the ecosystem level represents the most appropriate level of analysis for assessing the impacts on the functional components of the environment. It does not ignore the effects on the components of the ecosystem, but it also considers the important systemic effects.

2.2. Service providing units

The Millennium Ecosystem Assessment (MA, 2005) emphasizes the flows of ecosystem services, but disregards the stocks of resources that are essential for sustainability (Victor, 1991; Tomich et al., 2010). Apart from biophysical and chemical components, resource stocks consist of biodiversity components (MA, 2005). In order to explain the origin and the maintenance of ecosystem service flows an understanding of the link between biodiversity and ecosystem services is required (Haines-Young and Potschin, 2007a,b; Kremen et al., 2007). While research on the contribution of biodiversity to selected ecosystem processes is relatively well established (Vanderwalle et al., 2008), there is also a lack of a theoretical framework to link biodiversity with ecosystem services provision and human well-being (e.g. Balvanera et al., 2006; Carpenter et al., 2006; Diaz et al., 2006; Tilman et al., 2006). Research on linking biodiversity and ecosystem services has focused on the role that species and functional diversity (particularly in plants) play in modulating ecosystem processes such as primary production, nitrogen retention, decomposition and stability (Huston, 1997; Schwartz et al., 2000; Diaz and Cabido, 2001; Loreau et al., 2001; Tilman et al., 2001; Duffy, 2002; Srivastava and Vellend, 2005; Tilman et al., 2006). Approaches have been proposed that identify and quantify changes in ecosystem dynamics and their implications for ecosystem services (Luck et al., 2003, 2005; Haines-Young and Potschin, 2007a,b; Kremen et al., 2007). Luck et al. (2003) introduced the concept of “service-providing units” to explicitly link populations of species with the services they provide to humans. A service-providing unit can be defined as the component of biodiversity

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1 The concept of ecosystem is defined here following Jørgensen (2002) as a highly complex functional system that sustains life and includes all biological and non-biological variables. In this definition, spatial and temporal scales of ecosystems are not specified a priori but are based on the objectives of the analysis.
The ability of an ecosystem to provide multiple services can be determined by: i) allocating relevant ecosystem properties to each service and identifying which organisms or groups of organisms in the service-providing unit control these properties (Kremen, 2005), and ii) identifying the key characteristics (functional traits) and mechanisms by which these organisms affect ecosystem function (Luck et al., 2009). Given the systemic nature of ecosystems, a linear relationship between ecosystem services and functional traits is not expected, but it is possible to identify clusters that link functional traits and ecosystem services (de Bello et al., 2010). Each functional trait can play a role in different ecosystem processes and may be linked to several ecosystem services. The connections or associations between a trait and the related services will vary in amplitude and magnitude.

Recognizing the difficulties in applying the service-providing-units approach using species populations in real landscapes, Luck et al. (2003) suggested that the concept could include functional groups and ecological communities. Thus extended, the service-providing-units approach is potentially freed from traditional organizational hierarchies by defining any collection of individuals or species as a service-providing unit irrespective of their organizational level (Vanderwalle et al., 2008). Components of the service-providing unit are characterized by functional traits (i.e. their contribution to ecosystem processes). Functional traits are morphological, physiological or phenological traits that affect fitness via their effects on growth, reproduction and survival (Violle et al., 2007). Functional traits are strictly related to variations in rate functions and depend on the pattern of functional connections within a community (Violle and Jiang, 2009). Plant pests act as drivers of ecosystem change modifying the functional aspects of biodiversity components (i.e. functional traits) of the receiving environment that are part of one or more service-providing units. To better understand how IAS act on service-providing units, it is important to consider how they modify the functional traits and how these traits contribute to ecosystem services.

2.3. Trait-service clusters

The ability of an ecosystem to provide multiple services can be determined by: i) allocating relevant ecosystem properties to each service and identifying which organisms or groups of organisms in the service-providing unit control these properties (Kremen, 2005), and ii) identifying the key characteristics (functional traits) and mechanisms by which these organisms affect ecosystem function (Luck et al., 2009). Given the systemic nature of ecosystems, a linear relationship between ecosystem services and functional traits is not expected, but it is possible to identify clusters that link functional traits and ecosystem services (de Bello et al., 2010). Each functional trait can play a role in different ecosystem processes and may be linked to several ecosystem services. The connections or associations between a trait and the related services will vary in amplitude and magnitude.

Current knowledge on the functional traits underlying the delivery of ecosystem services across different trophic levels is still limited. Nonetheless, the recent increase in studies has enabled analyses of the relationship between biodiversity components and their role in ecosystem functioning (de Bello et al., 2008, 2010).

Service-providing units often comprise more than one species and there may be interspecific differences in the contribution to a given service. Species or populations may also contribute to more than one service or be antagonistic to the supply of a different service (de Bello et al., 2010) and this has to be remembered when one tries to identify the links between SPUs and services. The combination of functional traits across trophic levels controls the provision of multiple ecosystem services and can be used to identify trait-service clusters (Díaz et al., 2007; Kremen et al., 2007; de Bello et al., 2008, 2010). Also the spatial dimension has to be considered in the analysis of trait-service clusters. The strength of trait-service associations might depend on the spatial scale over which the effect of traits operate and the scale over which services are delivered.

2.4. Ecosystem services

The definition of ecosystem services is a debated issue, particularly with regard to the separation between benefits and the ecological structure and processes that form the basis for ecosystem services (MA, 2005; Boyd and Banzhaf, 2007; Fisher et al., 2008). Three important aspects in these definitions of ecosystem services are relevant for our purposes: a) ecosystem services must be ecological phenomena, b) they are natural structures/processes that are services only in relation to human interests, and c) they do not have to be directly utilized. Ecosystem services emerge as the functioning of certain structures under certain environmental conditions. This underlines the importance of both organization and structure, as well as processes and/or functions if they are used. Functions or processes become services if they yield benefit for humans.
Several classification schemes for ecosystem services have been proposed (e.g., in Wallace, 2007), starting from the one elaborated in the context of the Millennium Ecosystem Assessment (MA, 2005). However, in this paper we have used the scheme originally proposed by the Millennium Ecosystem Assessment (MA, 2005) because it has a broad content (Costanza, 2008) and is widely recognized and adopted. Following Carpenter et al. (2009) and de Bello et al. (2010), the regulating and supporting ecosystem service categories identified by the Millennium Ecosystem Assessment (MA, 2005) have been combined, because a clear objective distinction between the two categories is not available (Brauman et al., 2007; Carpenter et al., 2009; de Bello et al., 2010). Cultural services have not been considered here since they were outside the scope of our study.

The impact of IAS on ecosystem services has begun to attract the attention of both ecologists and risk assessors (e.g. Vilà et al., 2010). A list of the ecosystem services adopted for the ERA of plant pests is given in Table 1. Many of the ecosystem services in Table 1, and particularly those for regulating and supporting services, can be regarded more properly as ecosystem processes and not services per se (Wallace, 2007; Fisher et al., 2009). However, the analysis of ERA is simplified and can usefully be conducted by considering both the ecosystem services sensu stricto and the ecosystem processes that create the ecosystem services instead of assessing only “the benefits people obtain from ecosystems” (MA, 2005).

3. Assessment scheme

3.1. Scenario development and assumptions

The proposed assessment scheme applies to the ERA for invading or potentially invading plant pests. In most ERAs, qualitative/quantitative estimation of the impact cannot benefit from a deeper understanding of the ecological processes and their reaction to the driving force (Carpenter et al., 2009). The use of a scenario can help to address the complexity and uncertainty characterizing the ERA of plant pests. Scenario exercises are seen as particularly useful to assess future developments within complex and uncertain systems, such as ecosystems. As a consequence, scenarios have been widely used in ecosystem assessment issues (IPCC Watson RT and Core Writing Team, 2001; UNEP, 2002; EEA, 2005). In MA (2005), scenarios are defined as “plausible and often simplified descriptions of how the future may develop, based on a coherent and internally consistent set of assumptions about key driving forces and relationships”. Many other definitions of scenario have been proposed, but nearly all definitions have in common that scenarios explore a range of plausible future changes and that scenarios are neither predictions nor forecasts; scenarios are also not attempting to identify the most likely future trends (Zurek and Henrichs, 2007; Henrichs et al., 2010).

The purpose of the present framework is to develop and analyze scenarios that explicitly combine qualitative and quantitative information and estimates (EEA, 2001). In the absence of complete information or models assessing the effects of plant pests on ecosystems, a useful contribution can still be made by describing the causal chain by which an effect can be estimated. Even a narrative description of the development of impacts is an advance over having no information at all (Carpenter et al., 2009). More precisely, we are interested in producing scenarios related to environmental risk associated with plant pests. Scenarios are attempts to explore what future developments may be triggered by a driving force, in this case an exogenous driving force, i.e. a driving force that cannot or can only be partially influenced by decision makers (Henrichs et al., 2010). In scenarios most of the work is based on qualitative evaluations that can be translated into quantitative assumptions on the final state of the system (Henrichs et al., 2010). In the probabilistic evaluation at the basis of scenario development, expert opinions and assumptions are fundamental (Hein et al., 2006).

To develop scenarios, information on the initial condition of the system (e.g. the current distribution of the plant pest and the distribution of its host plants) and assumptions on future trends are required. These assumptions also include features that can modify or mitigate the degree of change in functional traits due to the pest and then in the ecosystem services level of provision (e.g., management activities, ecosystem resistance and resilience). The consideration of different configurations of assumptions leads to just as many scenarios to be developed. Also the consideration of critical uncertainties may require the development of additional scenarios (Henrichs et al., 2010).

The assumptions that provide the basis of a scenario exercise for an ERA of a plant pest are listed below.

3.1.1. Temporal horizon

The temporal horizon is critical and should be based on identifying a reasonable time period for the main issue of concern to be explored or managed (Henrichs et al., 2010) and the expected trends in evolution of the environmental impacts. There are three options for time horizon choice: i) a single time horizon, ii) multiple time horizons, and iii) a time horizon based on the worst case scenario when sufficient time has passed for the pest to invade the whole area of potential establishment. In the case of multiple time horizons, a scenario evaluation is required for each defined time horizon. There is a lack of clearly defined criteria to delimit a time horizon, but the following characteristics may assist in defining it:

a) The rate of spread of the pest in the PRA area, which has usually already been assessed in the full PRA scheme. The time horizon should be Related to the rate of spread: the higher the rate, the shorter the time horizon. The rate of spread depends on the biology of the pest (e.g. the number of generations per year, the rate of population growth, and the method of movement), the patterns of dispersal (e.g. whether random, stratified or human assisted), the continuity of suitable habitats and the climate.

b) The rate of appearance of the impact. The impact of an IAS may become apparent only after a lag phase (Aikio et al., 1994): the faster the appearance of changes in affected ecosystems the shorter the time horizon could be. Besides the biology and ecology of the pest, the time of appearance is strictly related to the system resistance (see below). Other factors influencing the lag phase are the ease of detection of the species, manifestation and visibility of symptoms and the adaptability of the pest. While some plant pests may take a very long time to cause impacts, mainly because the climate or due to competition or predation, pest risk managers still need an assessment of impacts under current conditions and the choice of a time horizon that is too long may make it difficult to justify phytosanitary measures.

<table>
<thead>
<tr>
<th>TYPE OF SERVICE</th>
<th>SERVICE</th>
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<tbody>
<tr>
<td>Provisioning services</td>
<td>Food</td>
</tr>
<tr>
<td></td>
<td>Fiber</td>
</tr>
<tr>
<td>Genetic resources</td>
<td>Biochemicals, natural medicines, etc.</td>
</tr>
<tr>
<td>Ornamental resources</td>
<td>Fresh water</td>
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<tr>
<td>Regulating and supporting services</td>
<td>Air quality regulation</td>
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<tr>
<td></td>
<td>Climate regulation</td>
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<tr>
<td></td>
<td>Water regulation</td>
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<td></td>
<td>Water cycling</td>
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<td></td>
<td>Soil formation</td>
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<tr>
<td></td>
<td>Erosion regulation</td>
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<tr>
<td></td>
<td>Nutrient cycling</td>
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<tr>
<td></td>
<td>Photosynthesis and primary production</td>
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<tr>
<td></td>
<td>Pest regulation</td>
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<tr>
<td></td>
<td>Disease regulation</td>
</tr>
<tr>
<td></td>
<td>Pollination</td>
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</tbody>
</table>
c) Process uncertainties. Over time, the assessments of impacts on ecosystems are affected by increasing uncertainty due to the unpredictable future pattern of action of other drivers (endogenous and exogenous) on ecosystem change. High levels of uncertainty in the mechanisms and effects involved in the interaction between the pest and the receiving environment indicate that a shorter time horizon is preferable.

3.1.2. Spatial scale
The spatial scale covers the area of potential establishment and corresponds to the total area potentially affected by the driving force (the pest) at the end of the selected time horizon. The spatial extent is determined by the selected time frame, the area of potential establishment and the assessment of the dynamics of spread in the risk assessment area. This information is usually provided in the related PRA. The assessment is performed for the area assumed to be affected. The worst case scenario is considered to be the area of potential establishment identified in the PRA document. The spatial scale also considers the grain (i.e. the spatial resolution) of the scenario. The purpose is to determine whether the impacts in the risk assessment area are likely to be spatially homogeneous and to assess the spatial variation in the magnitude of the impact.

3.1.3. Resistance
The impact of an IAS over time also depends on the capacity of the invaded ecosystem to resist ecosystem change. If we define impact as the percentage of loss in ecosystem service provision level, many different trends in the evolution of the impact can be expected. They may be considered between two different extremes: i) a system with low resistance the impact increases shortly after the introduction and rapidly reaches a maximum (Fig. 2A), and ii) in a highly resistant system the impact increases more slowly, (e.g. after a lag phase), and progressively increases towards the maximum level (Fig. 2B). However, in many cases the ecosystem’s resistance to the action of the driving force may result in a long (i.e. lag phase) or even permanent phase of low impact in the PRA area.

For the assessment of impact, a specific assumption of the resistance level of the invaded environment is required and the impact is evaluated accordingly. The selection of a particular level of resistance contributes to the choice of an appropriate time horizon.

3.1.4. Resilience
There are different hypotheses on the extent to which ecosystem resistance (Neuenschwander et al., 1987) can mitigate or reverse environmental impact. At the two extremes, the trend in ecosystem modification may be irreversible (low or no resilience) (Fig. 3A) or almost completely reversible (high resilience) (Fig. 3B). An assumption of the resilience of the invaded environment is required to evaluate the impacts and the strength and type of resilience need to be taken into account when setting an appropriate time horizon.

3.1.5. Management measures
Assumptions on the presence, type and efficacy of the existing management measures will influence the trends in impact. There are two options:

a) No management measures or actions are applied. The pattern of impact is not changed;

b) Management measures are applied that are capable of modifying the pattern of the impact through: i) containment and eradication, ii) modification of the intensity of the driving force action, and iii) improvement of the resistance and/or resilience of the invaded communities and ecosystems.

For the assessment, a specific assumption on management is required and the impact is evaluated accordingly.

3.2. Relating impacts to ecosystem service losses

3.2.1. Review of the literature
A review of the type and magnitude of the environmental impacts caused by the pest in the area of current and/or potential invasion (that may include the risk assessment area) is required. From this information, the ecological role and the ecological interactions that the pest may have in the current area of invasion can be defined. If the species has not invaded any other area or if the invasion is too recent and too little is known of its ecology in the invaded areas, the ecological role of the species can be evaluated from the situation in its native area and by using expert opinion. Although this information is generally a poor predictor of the potential impact on the risk assessment area, it may help to identify elements and mechanisms of the potential impact.

3.2.2. Identification of the service-providing units
Impacts on ecosystem services occur if the pest interacts with species or functional groups (populations or other functional units) that have implications for service provision. The components of the affected service-providing units are potentially countless. The evaluation must therefore focus on the principal species, e.g. potential host plants, or functional groups in the community substantially affected by the IAS. One can start from a minimum set of elements affected, i.e. the plant community or host plants affected by the plant pest. This minimum set can be expanded by considering other components of the ecosystem that are functionally related to these host plants. These could include herbivores associated with the plants, their competitors, pathogens and their vectors. In the plant health context, ecosystem services linked
to the functional traits of plant species that act as hosts or competitors of the plant pest are of primary interest.

3.2.3. Identification of the functional traits in the components of the service-providing units that can be positively or negatively affected by the pest

The potential modification of functional traits is evaluated assuming the presence of the driver. The functional traits for the service-providing units can be considered at three levels:

a) At the individual level. Many morphological, biochemical or regeneration traits of host plants related to growth, reproduction and survival can be modified by pests. For plants, easily measurable functional traits and standardized protocols for their assessment exist (Cornelissen et al., 2003). Such shortlists need to be further developed for other organisms and functional groups related to plants (e.g. pollinators).

b) At the population level. Properties, such as demographic traits and competitive capacities, that characterize the contribution of the host population to ecosystem processes can be modified by direct and/or indirect interactions with an IAS, resulting in changes in population abundance and distribution (Begon et al., 2006; Smith and Smith, 2006).

c) At the community level. Properties that characterize the contribution of a host community to ecosystem processes can also be modified under the pressure of an IAS. These factors have effects on fluxes of matter and energy, temporal and spatial dynamics, the evolution of biocoenoses, and on variation in stability and regulatory properties. Trait effects on ecosystem processes are mediated by the type, range and relative abundance of functional diversity (Díaz and Cabido, 2001; Loreau et al., 2001; Díaz et al., 2007) in a given community, the abundance and importance of dominant species in different trophic nodes, and the degree of functional dissimilarity in traits (Luck et al., 2003; Vanderwalle et al., 2008).

3.2.4. Trait-service clusters and type of effect

Given the systemic nature of social-ecological systems and the limited knowledge available on the ecological basis of ES, the identification of trait-service clusters (De Bello et al., 2010) is a complex task. Exploring the causal chain providing the unit, traits and services can result in a very complex pattern of connections and it is important to set limits to this exploration. We propose a heuristic approach to derive the most parsimonious set of relationships defining the trait-service cluster adapting it to the level of available knowledge and the resources available for the analysis.

Studies on the impact of IAS on ecosystem services (Vilà et al., 2010) and the analysis of trait-service clusters in different types of ecosystems (De Bello et al., 2010) can reveal recurrent aspects in the clusters, particularly if ecosystems in the same type of environments are considered (e.g. terrestrial ecosystems). The functional traits can be positively or negatively influenced by the pests and the variation in the traits can lead to positive or negative modifications of ecosystem services (Fig. 4). For each trait-service association an analysis of the types of effects has to be conducted.

3.2.5. Evaluation of changes in ecosystem services provision

For each trait-service association described for the service-providing units of interest an index is proposed to evaluate the intensity of the impact of trait modification on ecosystem service provision, given the assumptions in Section 3.1. Changes in ecosystem service provision occur due to direct impacts on target elements in the service providing unit with the subsequent indirect impacts on other components of the service providing unit.

For each affected ecosystem service, the relative magnitude of change (expressed in percentage of losses in the ecosystem service provision level) has to be estimated. The magnitude of the impact is categorized in the 5 classes listed in Table 2. Positive effects are not considered.

4. Assessment of the risks posed by Anoplophora chinensis to ecosystem services in Europe

The application of the proposed scheme is demonstrated with the example of the Citrus long-horned beetle, A. chinensis (Foster) (Coleoptera: Cerambycidae).

4.1. Life history, distribution and spread of A. chinensis

The Citrus longhorn beetle is a polyphagous plant pest, feeding on more than 70 plant taxa belonging to more than 20 different families of deciduous woody plant species including maple, birch, poplar, willow, apple, pear and citrus (Van der Gaag et al., 2010). Although all woody non-coniferous deciduous plants may be considered as potential hosts, complete development has not been confirmed in all tree species listed as hosts (Haack et al., 2010).

The beetle is native and widespread in Eastern Asia (Lingafelter and Hoebeke, 2002) causing extensive damage, especially since it can infest healthy trees. It was introduced to Europe around 2000 (Hérard et al., 2006; Haack et al., 2010; van der Gaag et al., 2010) and USA (with subsequent eradication; EPPO PQR, 2013). According to EPPO PQR (2013), the current situation in Europe is as follows: there have been recent detections and partly subsequent eradication of A. chinensis in Croatia (eradicating), Denmark (transient, under eradication), France (eradicating), Germany (eradicating), Guernsey (present, few occurrences), Italy (present, restricted distribution), Lithuania (no longer present), the Netherlands (eradicating), Switzerland (present, few occurrences) and United Kingdom (transient, under eradication). In Europe, this beetle poses significant threats to forestry, horticulture, ornamental trees and woodland.

Table 2

<table>
<thead>
<tr>
<th>Rating (J)</th>
<th>Magnitude of the impact, in %</th>
<th>Midpoint of the rating, in %</th>
</tr>
</thead>
<tbody>
<tr>
<td>Minimal</td>
<td>Zero or negligible</td>
<td>0</td>
</tr>
<tr>
<td>Minor</td>
<td>0 &lt; L ≤ 5%</td>
<td>2.5</td>
</tr>
<tr>
<td>Moderate</td>
<td>5 &lt; L ≤ 20%</td>
<td>12.5</td>
</tr>
<tr>
<td>Major</td>
<td>20 &lt; L ≤ 50%</td>
<td>35</td>
</tr>
<tr>
<td>Massive</td>
<td>L &gt; 50%</td>
<td>75</td>
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Fig. 4. The pest (driving force) can positively or negatively influence the functional traits of the service providing unit, which will lead to a positive or negative modification in ecosystem service provision.

Appendix C of EFSA, 2011.)
4.1. Natural spread

Based on the related species, Anoplophora glabripennis, that rarely flies further than 400 m (Dumouchel, 2004; Sacco, 2004) it is assumed that this species spreads slowly by natural means. However, at high population densities A. chinensis has been observed to fly more than 2 km (unpublished data referred to in Adachi (1990) without details) and its polyphagy increases the capacity for spread. Nevertheless, it is likely to take several years for populations to build up to high densities at new outbreak sites in Europe.

4.1.2. Human assisted spread

The import of host plants for planting from areas where A. chinensis is present, especially Acer spp. from China, is the most important pathway for introduction (Eyre et al., 2010; van der Gaag et al., 2010). A. chinensis may also be spread by vehicles. Spread with infested wood, though less probable, can also occur.

4.1.3. Impact

The Citrus longhorn beetle causes damage by larval tunneling in the cambial region (i.e. the tissues between the bark and wood), wood of the lower trunk and roots of its hosts. Both healthy and stressed trees (including bonsais) can be infested (Eyre et al., 2010; Haack et al., 2010). The disruption of vascular tissues after repeated attack with secondary fungal infestations may eventually lead to the death of the tree. Exit holes remain open for many years (Sabbatini Peverieri et al., 2012), leaving the tree vulnerable to intrusion by fungi and bacteria.

In China the greatest economic losses from A. chinensis in Asia have occurred in fruit tree plantations, where millions of trees have been killed, especially citrus. Outside their native range, tree mortality has also been recorded (Haack et al., 2010).

4.2. Development of the scenario

4.2.1. Definition of the service providing unit (SPU)

A. chinensis can cause impacts in various habitats, e.g. greenbelts, urban landscapes, gardens, orchards (e.g. citrus), and forests. The ecosystem services (ESs) these habitats provide allow them to be grouped into two SPUs: those in urban areas and those in more complex environments such as orchards and natural woodland. Impacts are likely to be greatest in semi-natural habitats and less evident in natural forests. However, since the information on the impacts in non-urban habitats is still very limited, we refer only to one SPU and the main focus in the assessment is put on the urban areas.

4.2.2. Projection of invasion

4.2.2.1. Definition of the temporal horizon. A. chinensis has a life cycle of 1–2 years in its native area and in Northern Italy (Adachi, 1994; Haack et al., 2010). It is probably even longer in northern Europe (Baker and Eyre, 2006; Van der Gaag et al., 2008) with a more limited capacity for natural spread (Adachi, 1990; Van der Gaag et al., 2010). This will result in both a slow population growth and a slow rate of continuous spread. Long distance spread occurs with human assistance by trading and transporting infested plants and infested wood. Since the pest is expected to take a relatively long time before significant environmental impacts are expected, the temporal horizon has been set at 30 years with a single time frame and a single scenario.

4.2.2.2. Definition of the spatial scale. Taking into account the area of potential establishment at the end of the selected time horizon, the spatial scale, is primarily limited by the pest's low rate of population growth and spread. It is assumed that less than 20% of the suitable area in which the SPUs are present will be affected. Its presence in the SPU will mainly depend on the mode of spread and the availability of host plants.

4.2.2.3. Estimation of resistance and resilience. Although several natural enemies have been identified in Lombardy (Delvare et al., 2004; Maspero et al., 2007; CABI, 2007), currently, the resistance of the ecosystem to A. chinensis is considered to be very low. In the long term resilience is expected to increase, partially regulating pest population abundance and reducing impacts due to the presence of natural enemies. These natural enemies are expected to stabilize pest population abundance and impacts to a lower level as the parasitoids adapt to the pest and their populations increase.

4.2.2.4. Main functional traits of the SPU affected by A. chinensis. At the individual level: larval feeding activity seriously affects tree physiology resulting in a decrease in growth and reproduction while the disruption of the tree's vascular tissues by the larval galleries causes structural weakness and physiological stress that can lead to tree death.

At the population level: almost all trees are at risk in urban areas, particularly because management is based on removal of infested trees. In non-urban areas the host population survival will depend on the high mortality rate.

At the community level: no selection is expected since the pest is polyphagous on deciduous broad-leaved trees but general stress is likely to occur due to modifications of the physical structure and primary productivity of the community, functional dissimilarity (niche diversity) and susceptibility to perturbation (e.g. in the stability of the soil community).

4.2.2.5. Management measures. Emergency measures have been adopted by the EU (European Commission, 2012) that require eradication in areas where the beetle has been found.

For the assessment, a specific management scenario is required. This describes the extent to which the measures are capable of modifying the pattern of the impact through: (i) containment and eradication and (ii) modification of the severity of pest impacts. In our scenario, it is evident that for (i), eradication is only possible at very early stages of infestation (thus in Lombardy, it seems no longer achievable), since insecticides are of low efficacy due to the hidden life stages of the beetle inside the tree and, although contact sex-pheromones have been identified, no long-range pheromones have been reported (Yasui et al., 2003). For successful eradication, all infested trees have to be felled and removed and all adults capable of reproduction killed (Van der Gaag et al., 2008). Containment may be easier to achieve but difficult to guarantee since the beetles may spread unnoticed. With regard to (ii) it is assumed that this is only possible by introducing biocontrol agents and these will have only medium efficacy with high uncertainty. Since, at present, the only effective method in the short term is to fell infested trees and this has a significant impact on ES, it is assumed that the impact of this management measures will in the long-term be reduced due to the expected future availability of biocontrol agents (Kaschio, 1996; Delvare et al., 2004).

4.3. Analysis of the impact on ES

4.3.1. Probability distributions of the reduction in ES provision levels

The assessment of the impact on ES provision level has been based on the scenario described in Section 4.2. In our scenario, population abundance is assumed to change over time and space according to population growth and spread. Population abundance is also under the influence of environmental forcing variables (abiotic factors), biotic factors related to the resistance and resilience and management measures that are or will become available within the time frame. The pest is considered to have the potential to cause high tree mortality, but the extent of this impact depends on population abundance. In this analysis we average the impact assessment over the temporal horizon of the 30 years. The scores on the magnitude of the impact have been assigned using a five level scale (Table 2), following the rating system in Appendix C of EFSA (2011). The probability distributions of
the expected reduction in the provision level of the affected ESs are described in Table 3. The variability in the scores takes into account the uncertainties in the magnitude of impact that are mainly related to the importance of resilience mechanisms and the availability and efficacy of management measures. Additional impacts on selected ESs, which are mainly related to the urban environment and to human health (food and fiber, ornamental resources and air quality), are described in Section 4.3.1.

4.3.2. Selected provisioning services affected by A. chinensis

4.3.2.1. Food and fiber. The feeding action of the larvae decreases the quality, volume and value of the timber, moreover secondary infestations by wood-destroying fungi have an even stronger harmful effect on the timber. An initial increase in the availability of firewood is expected but in the long term the availability of firewood will be reduced. A major impact on this service is expected only if A. chinensis has a similar impact on trees as A. glabripennis. However, since there is no supporting information in the literature, this has a high uncertainty. It is important to note that a high proportion of fiber production in the EU is from non-host plants (conifers) that are not included in the SPU we are assessing here.

4.3.2.2. Ornamental resources. Ornamental resources, especially trees, are susceptible to attack and can even be killed by the pest. This is the most seriously affected service, since, in addition to the damage caused by the pest, any findings lead to host destruction. The reduction in impact depends on the availability of management measures. A high impact is expected for this provisioning service, with a medium uncertainty. The level of uncertainty mainly depends on the effectiveness of natural enemies and biological control over time. Here, in addition, the rating is based on the proportion of ornamental host plants within the SPU.

4.3.3. Selected regulating and supporting services affected by A. chinensis

4.3.3.1. Air quality regulation. In urban areas ornamental trees contribute to the removal of pollutants. Here, a major effect is expected, with possible direct consequences for human health. For example, Donovan et al. (2013) found an association between tree loss from the spread of the emerald ash borer with increased mortality related to the cardiovascular and lower-respiratory systems. An effect on the level of ozone is also possible (Donovan et al., 2013). Lower impacts are expected in woodland. Therefore, a moderate impact is expected due to averaging of the impacts on urban (high impact) and woodland (low impact) areas, because of the consideration of only one SPU (see above). The uncertainty is rated as medium – not only because of the lack of information for woodland but also due to the difficulty in assessing the pest's role in the removal of pollutants.

4.3.4. Summarizing risks and uncertainties

The overall impact has been calculated according to the methodology provided in the rating system provided by Appendix C of EFSA (2011). For the provisioning services the overall risk is rated as high, but it is moderate for the regulating and supporting services (Fig. 6). The provisioning services are expected to be more affected by A. chinensis than the regulating and supporting services, in particular because of the impact on fiber and ornamental services (Fig. 6). Erosion regulation and air quality are the most affected regulating and supporting services.

The overall uncertainty has been calculated according to the methodology provided in the Appendix C of EFSA (2011) and is expressed in terms of the Shannon Entropy measure (from 0 to 100%). The overall uncertainties in the estimation of the impact of A. chinensis on the provisioning and the regulating and supporting services are both rated as medium (Fig. 7). The different ESs share the same sources of uncertainty, the most important being the control measures since it is difficult to predict how effective the control measures will be during the time horizon of 30 years and how the pests' population abundance will increase. The second source of uncertainty is related to the unknown responses (e.g., resistance and resilience) of the ecosystem. The third source of uncertainty is due to the possible differences between the impacts on urban and woodland ecosystems. Documented impacts on urban and peri-urban areas account for the lower uncertainties for ornamental resources and air quality regulation ESs (Fig. 7).

5. Discussion and conclusions

The IPPC Standards ISPMs 2 (FAO, 2008) and 11 (FAO, 2004) state that ERA should be included in PRAs. Several PRA procedures in line with IPPC standards that include the evaluation and prediction of environmental impact are available (e.g., USDA, 2012; Biosecurity New Zealand, 2006; Biosecurity Australia, 2007; EPPO, 2011; Kenis et al., 2012). However, no standardized methods are available to generate consistent, comparable and reproducible results. The new approach presented here differs from traditional approaches and enables to develop a scientific interpretation of the interface between natural processes and the way in which humans operate within ecosystems. The major differences compared to traditional methods are that our approach focuses on i) the effects of plant pests on ecosystem services, ii) the use of service-providing units and trait-service clusters, and iii) the development of a scenario for the assessment of the environmental impact of the invading organism.

The MA (2005) and several other studies (e.g., Vilà et al., 2010, 2011) have demonstrated the importance of ecosystem services and the way in which they are threatened by invasions. However, systematic assessments of impacts on ecosystem services are currently lacking. In particular, within PRA, the assessment of impacts of plant pests on ecosystem services has been neglected, though it has been shown in many different studies that plant pests can transform ecosystems and landscapes significantly and permanently (e.g., Vilà et al., 2010).

<table>
<thead>
<tr>
<th>Type of ES</th>
<th>ES</th>
<th>Probability distribution of the reduction in the ES on the level of the risk rating</th>
</tr>
</thead>
<tbody>
<tr>
<td>Provisioning</td>
<td>Fiber 0 or negligible</td>
<td>0 10 45 40 10 5</td>
</tr>
<tr>
<td></td>
<td>Ornamental 0</td>
<td>0 20 60 20 20 20</td>
</tr>
<tr>
<td></td>
<td>Primary production 0</td>
<td>0 40 30 30 0 0</td>
</tr>
<tr>
<td>Regulating and supporting</td>
<td>Air quality 0</td>
<td>0 40 40 20 0 0</td>
</tr>
<tr>
<td></td>
<td>Climate regulation 30</td>
<td>0 60 50 50 50 0</td>
</tr>
<tr>
<td></td>
<td>Water regulation and cycling</td>
<td>15 60 70 5 0 0</td>
</tr>
<tr>
<td></td>
<td>Erosion regulation 0</td>
<td>0 40 40 0 0</td>
</tr>
<tr>
<td></td>
<td>Nutrient cycling 5</td>
<td>5 40 40 10 0</td>
</tr>
</tbody>
</table>
In current ERA approaches, the focus is on the assessment of impacts on structural biodiversity at genetic, species, habitat, community, and ecosystem levels on a case by case basis. For the assessment of the functional components of biodiversity, which are evaluated by estimating how plant pests, that are acting as drivers of ecosystem change, modify the ecosystem services, no consistent procedures are available that capture the complexity and variety of mechanisms involved in the environmental impact of IAS. Our ERA for plant pests specifically addresses these mechanisms to provide an enhanced procedure for assessing the impacts of IAS on ecosystem services. It also expands the view on the consequences of introduction and spread of an IAS by explicitly determining the step from the detection of a change to the assessment of this change. The approach is based on a consistent and comprehensive framework of analysis and provides a solution to the problems related to the appropriate level (e.g., individual, population, community) and the type of analysis (e.g., trophic interactions, energetics, biogeochemical cycles, succession). Due to the detailed evaluation of impacts on both the (more) non-utilitarian value of nature considering the human influences on biodiversity as structural components of ecosystems on the one hand, and to the utilitarian perspective considering modification in functional traits relevant to ecosystem service provision on the other hand, the assessment of risks posed by plant pests to ecosystems and ecosystem services will be significantly enhanced. This will help decision makers to justify their measures on a more substantiated and comprehensible argumentation.

Our ERA method cannot be based on precise data, since it deals with non-measurable, complex structures, on which effects could be observed but in many cases not quantified. ERA in plant health is hampered by lack of data, a limited knowledge of the causal mechanisms leading to environmental consequences and the uncertainties inherent in extrapolating from the area where the pest is present to a new area. This lack of data and the difficulty to predict future developments is a common feature of all risk assessments. Since the mechanisms involved in environmental impacts of alien invertebrates, pathogens (Desprez-Loustau et al., 2007; Kenis et al., 2009) and plants (Vilà et al., 2011) are complex and various, each taxon would have to be studied...
separately through time-consuming and possibly costly field or laboratory studies. Such studies are not feasible within the framework, timescale and budget of a PRA, where a quick decision to determine whether a species represents a risk is needed. For most pests, therefore, the only feasible approach is to base the assessment procedure on expert judgment, i.e. the subjective view of the risk assessor including a risk rating, an uncertainty score and a written justification (Kenis et al., 2008), and the assessment of the potential environmental impact of a plant pest in a PRA area thus relies heavily on this expert judgment (Kenis et al., 2012). Therefore, one of the most important assets of our new approach is that it allows the explicit and structured assessment of complex systems and relationships; this is especially beneficial in situations where data are not available and where the uncertainties and complexity inherent to ecosystems and ecological processes exacerbate the difficulties of assessment.

To address the lack of data, an innovative methodology is needed that is able to analyze systemic effects consistently. Until now, these systemic effects have usually not been considered in PRA, since PRA traditionally looks at the narrow relationship between a pest and its host plant(s). Even if some indirect effects, such as the replacement of key-stone species or effects on species communities are assessed, risk assessors usually have limited information about indirect interactions. In our new procedure, interactions of environmental structures and processes within a systemic complex unit are considered at an appropriate ecosystem level. This helps the risk assessor to apply a broader view of the relationship between the (potentially) invading organism and the affected ecosystems.

Knowledge is still limited on the functional traits relevant to the provision of certain ecosystem services across different trophic levels, as well as on the causal mechanisms leading to environmental consequences. However, with this new approach it is possible to describe the causal chain through which the effects of IAS on ecosystem services can be anticipated. This is done by developing scenarios with defined, transparent assumptions, a defined time horizon and spatial scale, and take into account the resistance and resilience of the ecosystem to the invading species. Since the parameters are clearly defined and a standardized method is utilized, the approach is fully reproducible, enabling an integrated assessment to be made and impacts on the environment to be quantified. Such an approach has the potential to quantify the environmental impacts in economic terms. By integrating the service-providing units in the development of scenarios, populations of species can be explicitly linked with the services they provide to humans. The relationship between functional traits and ecosystem services is characterized by clusters of functional traits that are essential for each ecosystem service, where each functional trait can play a role in different ecosystem processes and may be linked to several ecosystem services. Due to the clear framework of the method, comparisons between assessments done with the same approach are also possible. The assessment of environmental impacts based on the ecosystem services concept is still a developing area and it is adapted to the current level of knowledge and the objectives that were identified when developing the framework. The example given for Anoplophora chinensis shows the application and practicability of the method and demonstrates how the impacts on ecosystem services can be rated and categorized, including the consideration of uncertainties. In this example, the main sources of uncertainty were the prediction of the effectiveness of control measures and the increase of the pest populations’ abundance, the level of resistance and resilience of the ecosystem, and possible differences between impacts on urban and woodland ecosystems. These are made transparent in the rating and show that the approach is also applicable when uncertainties are present. For further development, the method needs increasing application.

Since the framework proposed here is generally relevant to ecosystems, organisms therein and the related ecosystem services, this approach is applicable not only to plant pests but also to all kinds of organisms, for which impacts on ecosystem services are to be assessed. Therefore, it enables a comparable, reproducible and harmonized approach to assess effects on organisms on ecosystem services.

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