

Torymus sinensis: a viable management option for the biological control of *Dryocosmus kuriphilus* in Europe?

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Abstract The chestnut gall wasp *Dryocosmus kuriphilus* is a global pest of chestnut (*Castanea* spp). Established as a pest in the mid-twentieth century in Japan, Korea and North America, this species was first reported in Europe in 2002. Following the successful release of a biological control agent

Torymus sinensis in Japan, this parasitoid species has been released in Italy since 2005. Here we discuss the potential of *T. sinensis* as a viable management option for the biological control of *D. kuriphilus* in central Europe. We suggest that more consideration should be given to determining, (i) the conditions under which *T. sinensis* may attack alternative native gall wasp hosts and (ii) the likelihood of hybridization of this species with native *Torymus*. Both issues are central to predicting unassisted range expansion by released *T. sinensis*, and to assess the environmental risks associated with a more widespread release of this species in Europe.

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Handling Editor: Patrick De Clercq

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Keywords Alien species · Biological control agent ·
 Environmental risk assessment · Exotic species ·
 Invasive species · Non-target effects

Introduction

The introduction of biological control agents that are self-perpetuating, self-dispersing and potentially permanent is a way of controlling pest insects, and therefore an effective alternative to pesticides. Classical insect biological control has been successfully used for more than 120 years, and deployment of more than 2000 species of natural enemies has resulted in the control of at least 165 pest species worldwide (van Lenteren et al. 2006). Perkins (1897) was the first to describe non-target invertebrate

community changes following the release of coccinellid beetle biocontrol agents in Hawaii. Despite numerous early warnings, however, the non-target impacts of biological control agents were generally not seriously considered until the beginning of the 1990s (reviewed in Barratt et al. 2010). After the 1992 Rio Convention on Biodiversity, international organizations and national governments published general guidelines for the import and release of invertebrate biological control agents (e.g. The Food and Agriculture Organisation of the United Nations code of conduct for the import and release of exotic biological control agents; Greathead 1997). In Europe, a methodological guide on how to measure the environmental impact of using invertebrates for biological control of arthropods was recently published as a result of the Regulation of Biological Control Agents (REBECA) project funded by the EU-Commission (<http://www.rebeca-net.de/>). Prior to the release of a beneficial insect into the environment, all of the risks and benefits should always be weighed against each other (van Lenteren et al. 2003; Bigler et al. 2006; van Lenteren and Loomans 2006; De Clercq et al. 2011).

Here we provide a historical overview of the global invasion history of the chestnut gall wasp *Dryocosmus kuriphilus* Yasumatsu (Hymenoptera, Cynipidae), and the use of the parasitoid wasp *Torymus sinensis* Kamijo (Hymenoptera: Torymidae) as a biological control agent of this pest species. Using a new environmental risk assessment scheme proposed by van Lenteren and Loomans (2006) we examine the environmental risks associated with the release of *T. sinensis* in central Europe (Table 2, Fig. 1), discuss current gaps in knowledge and in the procedure, and highlight issues that should be urgently addressed by future research in order to ensure a comprehensive environmental risk assessment for the use of *T. sinensis*.

A historical overview

The pest *Dryocosmus kuriphilus*

Dryocosmus kuriphilus emerged as a pest in the mid-twentieth century and is now one of the most important global insect pests of chestnut (*Castanea* spp. Fagaceae; reviewed in Aebi et al. 2006). This pest

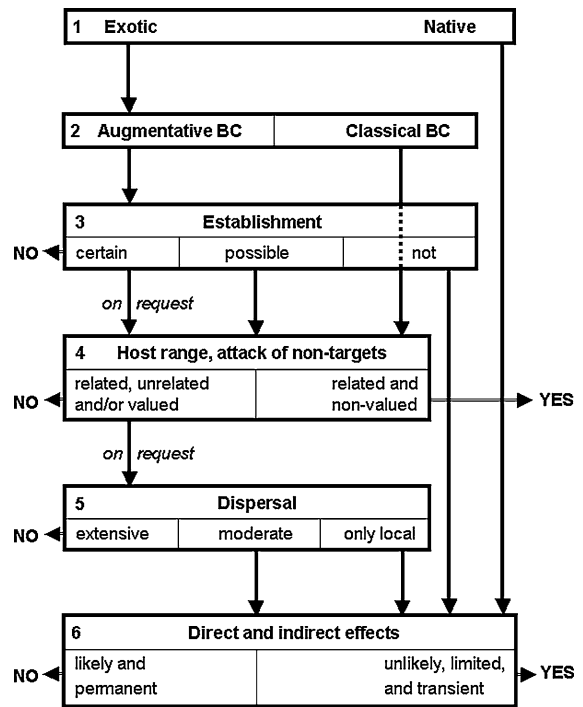


Fig. 1 Simplified scheme of an environmental risk assessment of an invertebrate biological control agent (after van Lenteren and Loomans 2006)

species disrupts growth by inducing gall formation on new shoots and leaves, suppressing nut production and causing a gradual decline in the vigor of these long-lived and slow-growing trees (reviewed in EFSA 2010). *Dryocosmus kuriphilus* is native to China, was accidentally introduced from there to Japan in 1941, and within 25 years it rapidly spread throughout Japan (Oho and Umeya 1975). In 1958, *D. kuriphilus* was recorded in Korea (at Chaenchun, Chungchungpuk-do) and over a period of 37 years this species spread across South Korea (reviewed in Aebi et al. 2006). *Dryocosmus kuriphilus* colonized North America from 1974 onwards, and was first recorded in Georgia (Cooper and Rieske 2007). By 1976, it was clear that *D. kuriphilus* was spreading in the USA, but little was published on the spread and impact of this species until Cooper and Rieske's overview in 2007. *D. kuriphilus* was reported from Nepal in 1999 (Abe et al. 2007), and from Italy in 2002 (Aebi et al. 2006). Introduction is thought to have been associated with import of eight Chinese chestnut cultivars to Piemonte between 1995 and 1996 (Aebi et al. 2006). Since 2002, *D. kuriphilus* has spread south through Italy to

Sicily, and more widely into south-eastern France and Corsica in 2005 (Aebi et al. 2006), Slovenia in 2005 (Knapič et al. 2009), Hungary in 2009 (Csóka et al. 2009), Switzerland in 2009 (Forster et al. 2009) and Croatia in 2010 (Matošević et al. 2010). Long distance dispersal of *D. kuriphilus* has been achieved so far via human-assisted movement of infested *Castanea* planting material (EFSA 2010). Mathematical models have estimated short distance dispersal by adult gall wasps at a rate of 8 km year⁻¹, with a variation comprised in a range of 3–12 km year⁻¹ (EFSA 2010), which is likely to assist the natural spread of this species to other areas of Europe. *Dryocosmus kuriphilus* could potentially spread throughout the range of *Castanea sativa* Mill. in Europe, but the areas currently considered most at risk from invasion by this pest species are northern Portugal, northern Spain and south-western France (EFSA 2010).

Options for controlling *D. kuriphilus*

Tight control of the movement of infested plant material will reduce long-distance dispersal of *D. kuriphilus* to new areas within Europe, but there are limited options available for managing existing *D. kuriphilus* populations and to reduce the magnitude of their impact and unassisted spread through European chestnut populations. Since the larval and pupal stages of *D. kuriphilus* are protected within their galls, conventional chemical control is regarded as largely ineffective (EFSA 2010). Developing resistant varieties of *Castanea* spp. could potentially be a viable management option, but this will only be beneficial for new planting and will not help existing chestnut populations (EFSA 2010).

Within some parts of their native range in China *D. kuriphilus* populations are kept at low densities, presumably by natural enemies, although little has been published and is known of alternative sources of mortality in this region (EFSA 2010). In Japan, South Korea, the USA and Italy the attack rates of indigenous parasitoid species are low (typically less than 2%; Stone et al. 2002; Aebi et al. 2007).

Torymus sinensis: use of this species as a biological control agent

Torymus sinensis is native to China and is the only Chinese parasitoid species of *D. kuriphilus* so far known to be host specific, and phenologically well

synchronized with *D. kuriphilus* (Moriya et al. 2003). *Torymus sinensis* females lay eggs into newly formed *D. kuriphilus* galls in early spring, and the parasitoid larva feeds externally on the mature host larva until pupation during late winter. In 1979 and 1981, a total of 260 mated *T. sinensis* females (reared from approximately 5000 *D. kuriphilus* galls imported from China) were released for biocontrol on Japanese chestnut trees at the Fruit Tree Research Station in Ibaraki prefecture (reviewed in Aebi et al. 2006). By 1989 this *T. sinensis* population had grown by 25 times and had become the most common parasitoid reared locally from *D. kuriphilus* (Aebi et al. 2006). After this release, *T. sinensis* dispersed successfully alongside expanding Japanese *D. kuriphilus* populations, and imposed effective biological control with the proportion of infested chestnut shoots falling well below the tolerable injury threshold of 30% (Gyoutoku and Uemura 1985). *Torymus sinensis* was also released in Georgia, U.S.A. in the late 1970s, where it tracked expanding *D. kuriphilus* populations into eastern North America and again reduced shoot infestation rates below the tolerable damage threshold, providing effective biological control (Cooper and Rieske 2007).

Following success in Japan and North America, preliminary studies on release of *T. sinensis* in Europe were conducted in 2003 and 2004 in Italy using imported Japanese *D. kuriphilus* galls (Quacchia et al. 2008). During these initial investigations a phenological mismatch between emergence of adult *T. sinensis* and local *D. kuriphilus* gall development (due to the temperature conditions experienced during shipment and rearing) meant that the parasitoids could not be released into the field. Instead they were used for behavioural trials, and these trials helped to improve later efforts in maintaining adults (Quacchia et al. 2008).

In 2005, more *D. kuriphilus* galls were imported from Japan and their development was slowed by artificial cooling. This enabled artificial synchronization of imported *T. sinensis* adult emergence with Italian *D. kuriphilus* populations, and in total ninety mated *T. sinensis* females were subsequently released at three infested *D. kuriphilus* field sites in Italy. Following successful establishment of *T. sinensis* at all three sites, a rearing program was set up to support the release of *T. sinensis* at additional infested sites in Italy, and *T. sinensis* is currently being introduced to

most regions across Italy, and to some parts of France (Quacchia et al. 2008).

Results collected from two of the Italian release sites (Robilante and Peveragno in the Cuneo province) indicate that the rate of parasitism of *D. kuriphilus* galls increased from less than 1% in the first year after release, to a mean of 16% by 2009 (Quacchia, unpublished data). However, it is still too early to assess the effectiveness of biological control by *T. sinensis* for long-term management of Italian *D. kuriphilus* populations. Successful control would constitute a reduction of gall wasp infection rates of less than 30% of chestnut shoots in the short term (Gyoutoku and Uemura 1985), and by a demonstration of population control by parasitization of *D. kuriphilus* in the longer term (Hassell et al. 1991). Data on density dependent attack rates of *D. kuriphilus* larvae or their spatial patterns, however, are currently unavailable for *T. sinensis* in Italy.

Assessing the risks of using *T. sinensis* as a biological control agent in Europe

A full environmental risk assessment relies on the identification and evaluation of potential risks associated with natural enemy release and the development of a plan to minimize them. The final step prior to release is to identify, assess and weigh up all adverse and beneficial effects in a risk-cost-benefit assessment (Bigler and K  lliker-Ott 2006). Here we apply the step-wise scheme proposed by van Lenteren and Loomans (2006) to assess the environmental risks associated with the release of *T. sinensis* in Switzerland and several other central European countries affected or threatened by *D. kuriphilus* (Table 2, Fig. 1). The factors used to evaluate an environmental risk assessment are listed in Table 1. Although all factors listed are of crucial importance for a comprehensive environmental risk assessment, there is growing awareness of the difficulties in obtaining relevant information for every aspect, and exhaustive study of the biology of a potentially beneficial arthropod can be very long and costly. The aim of the stepwise risk assessment procedure is to advise the release (or not) of a candidate biological control agent at an early step of the procedure, thus preventing unnecessary further assessment.

Table 1 Factors investigated in current environmental risk assessment schemes for biological control agents (BCA)

Establishment potential of the BCA
Dispersal of the BCA
Host range of the BCA
Direct effects of the BCA on other organisms in the ecosystem
Attack of non-target herbivores
Intraguild predation
Omnivory
Enrichment
Vectoring of pathogens
Indirect effects of the BCA on other organisms in the ecosystem
Competition
Intraguild predation
Apparent competition
Hybridization

Refer to van Lenteren et al. (2003) for definitions or details

The first question of the environmental risk assessment (Table 2, Fig. 1) concerns the origin and the intended use of the biological control agent. At step one, native and exotic biological control agents are differentiated. In the case of a release of *T. sinensis* in the invaded range of its host (Europe, United States and parts of Asia) we are dealing with an exotic species and are directed to step two. At step two, an augmentative biological control programme (where non-native establishment is not intended) is differentiated from a classical biological control programme (where long-term establishment is intended). In our case, classical biological control is the aim. In this case, the question of unwanted establishment is irrelevant and we are then directed to step number four. At step four, one needs to evaluate the host range of *T. sinensis* and decide whether it may attack non-target species. Previous work has shown that related oak gall wasps are attacked by similar parasitoid faunas (Bailey et al. 2009), so there is a general risk that *T. sinensis* could shift to native gall wasps related to *D. kuriphilus* (including native *Dryocosmus* species inducing galls on oaks). Current rearing data, though far from complete for alternative gall wasp hosts on other plants in China (see above) suggest that *T. sinensis* is specific to *D. kuriphilus*. It should be noted, however, that such apparent

Table 2 Schedule for an environmental risk assessment of *Torymus sinensis* in a certain area of release (after van Lenteren and Loomans 2006)

Step	Question—answer	Action
1	Origin—native	GO TO 6
	Origin—exotic, either absent OR present in target area	GO TO 2
2	Augmentative Biological Control (ABC) programme—establishment not intended	GO TO 3
	Classical Biological Control (CBC) programme—establishment intended	GO TO 4
3	Establishment unlikely	GO TO 6
	Establishment possible to very likely, risk threshold not crossed	GO TO 4
	Establishment possible to very likely, risk threshold crossed	No release
4	If monophagous OR if oligophagous/polyphagous AND only related AND no-valued non-targets attacked	Release
	If oligophagous/polyphagous AND related and unrelated non-targets attacked AND/OR valued non targets attacked	No release
5	Dispersal local (L = 1–2)	GO TO 6
	Dispersal outside target area (L = 3 or more) AND extensive (M = 2 or more) apply magnitude as a weight factor	
	If risk threshold is not crossed (ERI = 5 or less)	GO TO 6
	If risk threshold is crossed (ERI = 6 or more)	No release
6	Direct and indirect effects inside dispersal area unlikely (L = 1–2) AND at most transient and limited (M = 1–2)	Release
	Direct and indirect effects inside dispersal area likely (L = 3–5) OR permanent (M = 3–5)	No release

monophagy is exceptional among parasitoid species attacking cynipid galls (Bailey et al. 2009). If we accept the specialist status of *T. sinensis* as a given, the environmental risk assessment procedure concludes, by step four, that *T. sinensis* can be released in the framework of a classical biological control program against *D. kuriphilus*. However, two recent independent evaluations of *T. sinensis* as a candidate biological control agent identified several gaps in our knowledge of this species' biology: (i) knowledge of its non-target host range, and (ii) the risks of hybridization with native *Torymus* species (Aebi et al. 2011a; EFSA 2010). Each of these independent evaluations stressed that resolving these issues is essential for a comprehensive environmental risk assessment before further release of *T. sinensis* into other areas of Europe (Aebi et al. 2011a; EFSA 2010). We consider these issues in turn.

Potential for shifts to non-target hosts

The host range of *T. sinensis* has never been studied or tested in detail in either its native or introduced ranges, whether in the laboratory or field (Aebi et al. 2011a; EFSA 2010). To date, only one experimental

assay has been undertaken (Quacchia et al. 2008) and there are uncertainties about the experimental design and the choice of non-target host species (reviewed in EFSA 2010). During this assay, female *T. sinensis* were offered a limited range of alternative host galls comprising *Mikiola fagi* Hartig (Diptera: Cecidomyiidae), and the asexual generation of the oak gall wasps *Cynips quercusfolii* Linnaeus and *Andricus kollari* Hartig (Hymenoptera: Cynipidae). Lack of response by female *D. kuriphilus* to these alternative hosts (with no host inspection or host attack) led to the conclusion that no direct effects of release of *T. sinensis* would be expected (Quacchia et al. 2008). However, this range of alternative hosts is very limited, and other more logical alternative host galls on other plants (such as *Diplolepis* galls on roses, *Rosa*) were not considered.

The parasitoid communities associated with cynipid communities on oak, rose and chestnut galls are species-rich and generally regarded as closed—i.e. cynipids on each host plant taxon are attacked by relatively discrete (but slightly overlapping) sets of natural enemies (Csóka et al. 2005). Almost all of the parasitoids are chalcidoid wasps, and mortality is also inflicted by some inquiline gall wasps in the cynipine

tribe Synergini (Csóka et al. 2005; Ács et al. 2010). Although inquilines are highly specific to their host plant taxa, there is growing evidence from cynipid studies, and from studies on other insect invaders, that parasitoid species are able to switch between hosts attacking different host plants (Aebi et al. 2006). It is striking that 16 chalcidid parasitoid species from four families are known to exploit gall wasp species associated with more than one host plant taxon (Csóka et al. 2005). This list includes several *Torymus* species (*T. auratus* Müller, *T. flavipes* Walker and *T. scutellaris* Walker) as well as *Eupelmus* and *Eurytoma* species (Csóka et al. 2005). These observations suggest that although no alternative gall wasp hosts exist on native chestnut in Europe, the potential may exist for *T. sinensis* to exploit hosts on other plant taxa, such as cynipid galls on oaks, wild roses and herbs, all of which are attacked by *Torymus* species (Csóka et al. 2005; Askew et al. 2006).

The key issue for attack of non-target hosts is their seasonal phenology, and hence their availability for attack alongside *D. kuriphilus*. The EFSA Panel on Plant Health established a new alternative host species list for host-specificity testing of *T. sinensis*, which better matches the phenological characteristics of *D. kuriphilus* (EFSA 2010). All of the following are spring sexual generations (rather than the summer/autumn generations in the original test panel; see Quacchia et al. 2008): *Andricus curvator* Hartig, *A. inflator* Hartig, *Biorhiza pallida* Olivier and *Neuroterus quercusbaccarum* Linnaeus on white oaks (such as *Quercus robur* Linnaeus, *Q. petraea* (Matuschka) Liebl and *Q. pubescens* Wild.) and *A. cydoniae* Giraud, *A. grossulariae* Giraud, *A. lucidus* Hartig, *A. multiplicatus* Giraud, and *Dryocosmus cerriphilus* Giraud on semi-evergreen Cerris oaks (such as *Q. cerris* Linnaeus, *Q. suber* Linnaeus and *Q. trojana* Webb) (EFSA 2010; G. Csóka, pers. comm.). This set of species provides the closest phenological match to the flight period of *T. sinensis* females (i.e. between mid April and mid/end May in Piedmont, Italy). As the testing of more than ten species of non-target arthropods may be impractical and often unnecessary (Sands 1997), and since it is believed that a carefully designed host-specificity test on a few species related to the target species should provide adequate and reliable information (Sands 1998; Kuhlmann et al. 2006), we highly recommend

using this EFSA species list to perform a host-range assessment for *T. sinensis*.

If *T. sinensis* can be confirmed as a specialist parasitoid that does not attack unrelated non-target species, then it could be considered as a candidate for biological control of *D. kuriphilus* beyond its current range. By contrast, if *T. sinensis* displays a broader host range it would be considered too risky for release as a biological control agent into other areas of Europe. Consideration of *T. sinensis* population dynamics would undoubtedly improve the overall assessment of the probability of attack of non-target species. Demonstration of a broader host range would raise concern over the potential risks to native non-target species caused by an already released and growing population of *T. sinensis* in Italy.

What happens if *T. sinensis* shifts onto alternative native hosts?

Although the probability may be small, the possible impacts of such a shift onto native hosts—either through future releases of *T. sinensis* or population expansion by those already released in Italy and France—should be considered. Where chestnut gall wasps are abundant, there is a risk that high local population density of *T. sinensis* will suppress local populations of alternative hosts through apparent competition (Holt and Lawton 1994). Another possibility is that a future host shift by *T. sinensis* to other hosts might allow this parasitoid to outstrip range expansion by *D. kuriphilus*. While *D. kuriphilus* is likely to remain an essentially southern species in Europe due to the distribution of its chestnut hosts, alternative gall wasp hosts on oaks and roses are found much further north. As has been seen in oak gall parasitoids (Nicholls et al. 2010), a shift from galls on chestnut to those on alternative hosts would allow *T. sinensis* to disperse beyond the limits of sweet chestnut. We recommend that monitoring for presence of *T. sinensis* in non-target host populations should be maintained. Given the delay commonly observed in recruitment of parasitoids to non-native gall wasp hosts (Schönrogge et al. 1996), we recommend that monitoring should best focus on areas in Italy where population densities of *T. sinensis* are highest and have been longest established.

Potential for hybridization with other *Torymus* species

Hybridization of a biological control agent with native species is considered as an environmental risk to non-target species (Table 1, an indirect effect in Fig. 1), and is a general threat to native biodiversity from related introduced or invading species (e.g. Perry et al. 2002). In theory, insect species introduced for biological control may interbreed with native species. It is worth noting that the only reported case to date involves *T. sinensis* and a Japanese native *Torymus* species, *T. beneficus* Yasumatsu et Kamijo. Hybridization was suspected, and *T. sinensis* and *T. beneficus* were successfully crossed in the laboratory to produce fertile hybrid females (Moriya et al. 1992). Hybrids were also detected in the field and molecular markers proved their hybrid origin (Yara et al. 2000). Although *T. sinensis* is the only biological control agent shown to have hybridized with a native species, it is very likely that the proposed stepwise environmental risk assessment scheme outlined previously (Fig. 1), would allow its release into Europe without first evaluating the hybridization risks with native *Torymus* species. Sixteen native oak gall wasp parasitoids from four families have been consistently reared from *D. kuriphilus* (Aebi et al. 2006, 2007). Among them, three *Torymus* species (*T. auratus*, *T. flavipes* and *T. scutellaris*) have potential to hybridize with *T. sinensis* as these closely related species overlap geographically (they may even parasitize the same individual galls on the same chestnut trees), and probably also overlap in their seasonality. Examination of the potential for hybridization between *T. sinensis* and native congeneric species is urgently required so that the genetic integrity of native *Torymus* species developing on oak gall wasps can be protected.

Refining the environmental risk assessment for *T. sinensis*

One of the aims of the stepwise environmental risk assessment scheme described previously (Table 2, Fig. 1) is to reach a conclusion about whether (or not) to release a new biological control agent, without having to conduct long and costly experiments to answer all aspects of a classical environmental risk

assessment (Table 1). In the case of *T. sinensis*, however, using this scheme may result in premature release before the risks have been adequately assessed (as described earlier). To overcome this difficulty, we suggest using the environmental risk assessment scheme developed by the EU-funded project Evaluating Environmental Risks of Biological Control Introductions into Europe (ERBIC; van Lenteren et al. 2003) to identify and weigh all potential environmental risks associated with further release of *T. sinensis*. The ERBIC environmental risk assessment scheme is based on the identification and evaluation of all potential negative effects associated with the release of a biological control agent that can be named and measured. Then the probability that these effects will occur is evaluated. The risk of negative effects is the product of the likelihood (probability) of occurrence multiplied by the magnitude of impact. Table 3 describes qualitative scales and numerical values for a risk's probability and magnitude. As in the previous environmental risk assessment scheme the following ecological determinants of risks are considered: establishment, dispersal, host range, direct and indirect effects. A description of probability and magnitude for establishment, dispersal, host range as well as direct and indirect effects is given in Tables 4 and 5. An overall risk index is calculated by multiplying the values obtained for the probability and the magnitude of all potential risks individually and by summing the values obtained for establishment, dispersal, host range and direct and indirect effects. One of the criticisms of the ERBIC environmental risk assessment scheme is that numerical values do not allow clear separation between risk categories. This may in turn lead to misinterpretation or even manipulation of the data. To overcome this problem, weighted factors were included in the risk level calculation (van Lenteren and Loomans 2006). Here, for simplification and given that risk indices could not be calculated for all environmental risks we followed the ERBIC environmental risk assessment scheme. Environmental risk indices may then vary between 5 (5 times, 5×1) and 125 (5 times, 5×5). A low risk category (below 35) allows a proposition to release the biological control agent. An intermediate risk category (between 35 and 70 points) justifies a request for additional information on certain aspects of the environmental risk assessment by the regulators. High risks (more than 70 points) lead to a

Table 3 Qualitative scales and numerical values for probability (a) and magnitude (b) of environmental negative impact of a BCA (after van Lenteren et al. 2003)

	Description	Numerical value
(a) Probability		
Very unlikely	Not impossible but only occurring in exceptional circumstances	1
Unlikely	Could occur but is not expected to occur under normal conditions	2
Possible	Equally likely or unlikely	3
Likely	Will probably occur at some time	4
Very likely	Is expected to occur	5
(b) Magnitude		
Minimal	Insignificant (repairable or reversible) environmental impact	1
Minor	Reversible environmental impact	2
Moderate	Slight effect on native species	3
Major	Irreversible environmental effects but no species loss, remedial action available.	4
Massive	Extensive irreversible environmental effects	5

Table 4 Description of probability for establishment, dispersal, host range, direct and indirect effects (after van Lenteren et al. 2003)

Establishment ^{a,b} in non-target habitat	Dispersal ^c potential (m)	Host range ^d	Direct ^a and indirect effects
Very unlikely	<10	0 species	Very unlikely
Unlikely	<100	1–3 species	Unlikely
Possible	<1000	4–10 species	Possible
Likely	<10000	11–30 species	Likely
Very likely	>10000	>30 species	Very likely

^a As in Hickson et al. (2000)

^b The propensity to overcome adverse conditions (winter or summer: physical requirements) and availability of refuges

^c Distance moved per release (take number of generation per season into account); determine dispersal curve, sampling at points at 10, 100 and 1000 m, sampling period is 50% life-span

^d The propensity to realize its ecological host range in the release area

proposition not to release the biological control agent. Results of this approach applied to *T. sinensis* as a biological control agent of *D. kuriphilus* in Italy are summarized in Table 6. *Torymus sinensis* is known to have successfully established in Italy (Quacchia et al. 2008), and therefore based on these data, the ‘establishment likelihood’ of this biological control agent in other areas of Europe is evaluated as very likely (with a numerical value of 5). The lack of data on the potential for non-target attacks by *T. sinensis* prevents us from evaluating the ‘establishment magnitude’ (i.e. the percentage of potential non-target habitat where the biological control agent may establish) of this species. The ‘dispersal probability’ (i.e. the distance

moved per release, per generation) of *T. sinensis* was considered to be less than 10000 m (after Moriya et al. 1989; Moriya et al. 2003) and assigned a numerical value of 4. Given the high recapture rate obtained in Italy (Quacchia et al. 2008) the ‘dispersal magnitude’ (i.e. the percentage of released biological control agent dispersing from the target release area) was considered to be much higher (25%) and was therefore assigned a value of 5. The lack of detailed data on the host range of *T. sinensis* does not allow us to evaluate the propensity of *T. sinensis* to realize its ecological host range in the release area (probability) or the taxon range that *T. sinensis* actually attacks (magnitude). In the case of *T. sinensis*, hybridization

Table 5 Description of magnitude for establishment, dispersal, host range, direct and indirect effects (after van Lenteren et al. 2003)

Magnitude	Establishment ^a in non-target habitat	Dispersal ^b potential (%)	Host range ^c	Direct ^d and indirect ^e effects
Minimal	Local (transient in time and space)	<1	Species	<5% mortality
Minor	<10%	<5	Genus	<40% mortality
Moderate	10–25%	<10	Family	<40% mortality and/or >10% short term population suppression
Major	25–50%	<25	Order	>40% short term population suppression, or >10% permanent population suppression
Massive	>50%	>25	None	>40% long-term population suppression or local extinction

^a Percentage of potential non-target habitat where biological control agent may establish

^b Percentage of released biological control agent dispersing from target release area

^c Taxon range that biological control agent attacks

^d Direct effect: mortality, population suppression or local extinction of directly affected non-target organisms; see Lynch et al. (2001) for details

^e Indirect effects: mortality, population suppression or local extinction of one or more species of non-target species that are indirectly influenced by the released biological control agent

Table 6 Risk indices for the release of *T. sinensis* in central European chestnut forests (based on current knowledge)

Criterion	Probability (P)	Magnitude (M)	P × M
Establishment	5	n.a.	n.a.
Dispersal	4	5	20
Host range	n.a.	n.a.	n.a.
Direct and indirect effects	3–5	1	3–5

n.a. data not available

was considered as possible to very likely (value of 3–5) even if its magnitude (mortality, population suppression or local extinction of non-target species) was considered minimal (value of 1).

Results from Japan, North America and Italy indicate the potential for *T. sinensis* to be a viable management option for the biological control of *D. kuriphilus* in Europe. As described above, however, insufficient data are currently available to perform a detailed, accurate risk assessment for the use of *T. sinensis* as a biological control agent.

Conclusion

By adopting a combined approach and utilising both the environmental risk assessment scheme proposed by van Lenteren and Loomans (2006; Table 2, Fig. 1) and the ERBIC project (van Lenteren et al. 2003) we have used current knowledge to identify all of the

potential environmental risks of using *T. sinensis*, and highlighted where future research effort should be targeted to better inform the environmental risk assessment. A better formulated risk assessment would help to determine whether the risk of introducing *T. sinensis* outweighs the risks associated with the use of other control options (e.g. chemical control). We strongly argue that more consideration should be given to determining (i) the conditions under which *T. sinensis* could target alternative hosts (life table analyses would greatly improve assessment of the consequences of non-target host use), and (ii) the likelihood of hybridization with native *Torymus* species. In the case of *T. sinensis*, we recommend first performing host specificity tests before spending large amounts of time performing costly hybridization tests, because evidence of attack of native oak galls would be sufficient to reach a decision not to release *T. sinensis*. It is, however, important to note that host specificity tests only explore direct effects,

and indirect effects (such as competitive resource depletion, shared hosts and natural enemies; Holt and Lawton 1994) are also crucial criteria to evaluate (but often are difficult to establish; Schönrogge and Crawley 2000). At present, however, these indirect effects are not considered under the current environmental risk assessment scheme. In addition, other factors such as host species used, host location behavior, location of the gall on the host plant and phenology also need to be considered as they can also influence the outcome and reliability of host specificity tests (Bailey et al. 2009).

Ecological processes such as intraguild predation (see chapter Aebi et al. 2011b), enrichment, apparent competition and hybridization are very complex and require extremely detailed further studies. Recruitment of native oak or rose cynipid parasitoids to the chestnut community may cause inadvertent changes to community richness, and detailed monitoring of the communities associated with both native cynipids and *D. kuriphilus* are therefore also required. Overall, current evidence suggests that release of *T. sinensis* could have a wide range of potential impacts, and failure to consider these before further release of *T. sinensis* into Europe is unwise.

Acknowledgments This work was supported by an EHFI NERC grant NE/H000135/1 to KS, GNS and MG. We thank Franz Bigler for his valuable comments on the manuscript.

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