The suitability of weed risk assessment as a conservation tool to identify invasive plant threats in East African rainforests

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A R T I C L E   I N F O

Article history:
Received 4 November 2008
Received in revised form 7 January 2009
Accepted 13 January 2009
Available online 24 February 2009

Keywords:
Biological invasions
Disturbance
Exotic
Agroforestry
ROC

A B S T R A C T

We tested the ability of an international Weed Risk Assessment (WRA) protocol to predict invasion status of 230 alien plant species introduced via a botanical garden to tropical rainforest in Tanzania. The reliability and accuracy of WRA in discriminating between invaders and non-invaders was independently assessed using field data on species demography and distribution. The WRA rejected 83% of known invaders, and accepted 74% of non-invaders. Only 1% of accepted species were known invaders at the site. WRA performance varied among different growth forms, and underestimated the risks arising from palm species. Among those species that had naturalised, the WRA was better at identifying invaders of open rather than forest habitats. The WRA score was significantly correlated with how widespread species had become at the site, suggesting some capacity to predict spatial spread at a landscape scale. Knowledge of propagule pressure and residence time did not increase explanatory power. These results indicate that the WRA was able to discriminate between invaders and non-invaders with accuracy comparable to similar assessments in temperate and sub-tropical regions. It could be made more effective by weighting traits important in tropical forests e.g. certain growth forms, shade tolerance etc. more heavily. Such a modified WRA could be used successfully elsewhere in the palaeotropics as a screening tool to identify the risk of invasion arising from plants introduced for agroforestry, horticulture or landscaping. Given the increasing pressures on tropical forests and importance of agroforestry to local economies, the WRA protocol represents a useful conservation tool.

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1. Introduction

The environmental and economic costs of alien plant invasions are widely appreciated (Cronk and Fuller, 1995; Pimentel et al., 2001, 2005), and the most serious plant invaders can threaten the conservation of native vegetation and plant species (Rouget et al., 2003; Jager et al., 2007; Ens and French, 2008; Gerber et al., 2008). Many agroforestry species have been introduced and disseminated in developing tropical countries to halt land degradation, improve livelihoods and ease resource-use pressure on natural vegetation of conservation value, yet some have become invasive (Richardson et al., 2004). It is generally accepted that the most cost-effective method of curbing the consequences of biological invasions is to prevent the introduction of alien plant species that have a high risk of becoming invasive, whilst simultaneously allowing entry of species that are low risk, and may be of social and economic value (Hulme, 2006; Keller et al., 2007). Differentiating the characteristics that define these two groups of species is a significant challenge within invasion biology (Rejmánek et al., 2005). However, Pheloung et al. (1999) developed the Australian Weed Risk Assessment (hereafter A-WRA) as a weed screening system based on 49 questions that relate to the biogeography, invasion history, traits and ecology of the target species. The A-WRA has become the most widely applied and consistently tested scheme that quantifies plant invasion risk and allows a decision to be made whether to ‘reject’, ‘accept’ or ‘further evaluate’ a species (Gordon et al., 2008). The WRA has been successfully integrated into national policy in Australia and New Zealand, and has been shown to have net economic benefits when applied (Keller et al., 2007). Daehler et al. (2004) further developed the A-WRA specifically for the tropical ecosystems of Hawaii and the Pacific Islands (hereafter H-WRA). The resulting H-WRA has the distinction of a more strongly tropical focus in the climate and soils questions intended to be geographically adjusted (Gordon et al., 2008), as well as a secondary screening step that assists in reducing the number of species requiring further evaluation.
Nevertheless, both the A- and H-WRA suffer from several limitations. First, a weak but significant correlation has been found between a species individual WRA score and the number of questions answered in the tested H-WRA (Daehler et al., 2004), and therefore poorly studied species could conceivably receive a spurious low score for invasion risk. Second, the WRA score is strongly influenced by whether a species has a history of invasion elsewhere (Caley and Kuhntert, 2006; Weber et al., 2009). Although a useful indicator of potential invasive behaviour (Scott and Panetta, 1993; Reichard and Hamilton, 1997), it is only valuable if a species has a history of introduction outside its native range. However, a significant proportion of invasive weeds may have no history of introduction elsewhere. For example, 20% of recently naturalised weeds in New Zealand have no prior record of invasion (Williams et al., 2001). Third, successful invasion can depend on the effects of propagule pressure (Lockwood et al., 2005) and time since introduction (Wilson et al., 2007). These two factors are purposely not included in the WRA which was developed as a screening tool for new species, and this could potentially help to explain inconsistencies in the classification of invasion risk.

In this study, we used Amani Botanical Garden (hereafter ABG) in the East Usambara mountains of Tanzania to assess, for the first time, the utility of H-WRA in a continental tropical forest ecosystem. The location presents a unique opportunity to test and advance our understanding of weed risk assessment since:

1. Several hundred species were introduced to the ABG in the early 20th Century and the extent of their subsequent naturalisation and spread has been recently assessed (Dawson et al., 2008). Thus we were able to compare H-WRA scores with actual objective measures of invasion rather than more subjective expert scores of invasiveness (Gordon et al., 2008a).
2. Previous A- and H-WRA analyses have been undertaken at the regional or continental scale, whereas this study represents the first assessment at a landscape scale with the advantage that analyses can be refined to assess risks for individual habitats such as plantations, forest edges and closed canopy forest.
3. Detailed planting records enable the influence of residence time and introduction effort on invasion to be assessed (Dawson et al., 2009).

If the H-WRA system can accurately identify the most invasive plants at this location, it could be applied more widely as a pre-introduction screening tool for other tropical continental areas and botanical gardens. This study is particularly pertinent given recognition within the agroforestry sector that standard protocols to assess invasion risk of potential species are required (Richardson et al., 2004). As the potential benefits of agroforestry extend to ecosystems. Nevertheless, both the A- and H-WRA suffer from several limitations. First, a weak but significant correlation has been found between a species individual WRA score and the number of questions answered in the tested H-WRA (Daehler et al., 2004), and therefore poorly studied species could conceivably receive a spurious low score for invasion risk. Second, the WRA score is strongly influenced by whether a species has a history of invasion elsewhere (Caley and Kuhntert, 2006; Weber et al., 2009). Although a useful indicator of potential invasive behaviour (Scott and Panetta, 1993; Reichard and Hamilton, 1997), it is only valuable if a species has a history of introduction outside its native range. However, a significant proportion of invasive weeds may have no history of introduction elsewhere. For example, 20% of recently naturalised weeds in New Zealand have no prior record of invasion (Williams et al., 2001). Third, successful invasion can depend on the effects of propagule pressure (Lockwood et al., 2005) and time since introduction (Wilson et al., 2007). These two factors are purposely not included in the WRA which was developed as a screening tool for new species, and this could potentially help to explain inconsistencies in the classification of invasion risk.

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2. Methods

2.1. Study area

The study was undertaken in the lowland and submontane rainforests of the East Usambara mountains in northeast Tanzania, one of the most valuable conservation areas in Africa and constituting one of the world centres of plant diversity (Sayer et al., 1992). The ABG (5°05’30”S, 38°38’10”E) sits in one of the largest remaining forest reserves in the East Usambaras and was formally established in 1902 (Iversen, 1991). More than 500 species (mostly woody) were planted over a 28-year period in a series of trial plantations spread over some 300 ha. The ABG originally consisted of 20 plantation blocks, divided into 141 compartments, varying in shape and size from 0.1 to 7 ha, and contained almost 2000 species plots (Dawson et al., 2008). Management largely ceased in the 1960s (Iversen, 1991), and currently one third of species (214) are still present in the ABG (Dawson et al., 2008).

2.2. The weed risk assessment system

The H-WRA protocol (Daehler et al., 2004) was used with minor modifications to two questions relating to fire hazard since the ABG climate is humid with rainfall throughout the year, and fires are not known to usually occur (Iversen, 1991). Thus question 4.08 (‘creates a fire hazard in natural ecosystems’) was answered ‘no’ for all species and ‘fire’ was omitted from question 8.04 (‘tolerates, or benefits from mutilation, cultivation or fire’). Climate matching is an essential component of the H-WRA and whilst the use of climate matching software has been advocated (Pheloung et al., 1999), other proxies or default scores have generally been used (Gordon et al., 2008a). We generated systematic scores for the climate match questions based on information on the species native latitudinal range. For question 2.01 (species suitable to tropical or subtropical climates), species with a latitudinal range midpoint between 20° North and 20° South (i.e. centred on the tropics), were given a score of two; those with midpoints between 20° and 30° North or South (corresponding to ‘subtropical’) were given a score of one, and species with midpoints >30° North or South (i.e. ‘temperate’) were scored as zero. To address question 2.02 (quality of climate match data), a score of two was given for species that had a published latitudinal range, a score of one if the range was described but latitudes had to be obtained from atlases, and if the range was uncertain, the score was zero. Standard protocols for answering all other questions were followed across all species (Gordon et al., 2008b).

Following Pheloung et al. (1999), a minimum of 10 questions had to be answered for each species, with two in the biogeographic section, two in the undesirable traits subsection, and six of the remaining questions for biology/ecology. The H-WRA scores were calculated overall as well as agricultural and environmental scores, which were calculated using questions relevant to open agricultural habitats and natural areas (environmental), respectively, following Pheloung et al. (1999). As in other assessments, standard thresholds of risk were adopted such that species with a final overall score above six were rejected, species with a score of 0 or less were accepted, and a score of 1–6 meant the species required further evaluation (Pheloung et al., 1999). Species identified as requiring further evaluation were subsequently processed through a secondary screening that uses a subset of WRA questions in a decision tree (Daehler et al., 2004). Information sources used to answer questions included the peer-reviewed literature, African floras, online and CD ROM databases such as the Electronic Plant Information Centre (Royal Botanic Gardens Kew, 2008) and the Forestry Compendium (CAB International, 2005), online invasive species
databases, and individual species reports and fact sheets (ecological and horticultural). In addition, published H-WRA protocols were available for 91 species (Daehler, 2008) and this information was adapted or modified where appropriate for the ABG.

2.3. Measuring invasion status

Following invasion categories in Dawson et al. (2008), 214 species with known introduction history in ABG were placed in one of the following classes:

1) Surviving \( \text{(} n = 113 \text{)} \). Original plants present, but no seedlings or saplings.
2) Regenerating \( \text{(} n = 41 \text{)} \). Seedlings and saplings present, but not new adults. This category is equivalent to ‘casuals’, as defined by Richardson et al. (2000a) and implemented by Krivánek and Pyšek (2006).
3) Naturalised \( \text{(} n = 44 \text{)} \). Newly recruited adults found, but in 10 or fewer new compartments.
4) Spreading \( \text{(} n = 16 \text{)} \). Naturalised and found in >10 new compartments; species spreading only vegetatively must be found in at least one other new compartment to be in this category.

In addition, 16 species with unclear introduction records were classified with nine described as naturalised and seven as spreading (found in >10 compartments, or were known to be widespread both within the botanical garden and in the wider nature reserve). We successfully completed the H-WRA for 208 species (90%) and the distribution among classes was: 101 surviving (88%), 37 regenerating (90%), 47 naturalised (89%), and 23 spreading (100%). Naturalised and spreading classes each included seven species with unclear planting history.

Using systematic surveys of the ABG as well as the submontane and lowland rainforest adjacent to the plantations, we grouped naturalised and spreading species into four habitat categories:

1) Plantation \( \text{(} n = 17 \text{)} \). Species that were naturalised solely in overgrown plantations, and were not observed in open areas or in adjacent semi-natural forest.
2) Open \( \text{(} n = 14 \text{)} \). Species that may be in plantations but were found in open, disturbed areas, roadsides, cultivated and grazed land, but not in forest areas.
3) Forest edge \( \text{(} n = 22 \text{)} \). Species had colonised forest edges and forested areas, but were not established as adults in semi-natural forest >150 m from edges.
4) Forest \( \text{(} n = 17 \text{)} \). These species were found establishing as adults in semi-natural and natural forest >150 m from edges.

2.4. Analysis

We used ANOVA with post-hoc Bonferroni-corrected pairwise comparisons to analyse the relationship between H-WRA score and (a) invasion status, (b) habitat for naturalised and spreading species, and (c) growth form. For habitat comparisons, we considered the overall H-WRA scores as well as the separate environmental and agricultural scores. We calculated the accuracy and reliability of the overall H-WRA predicted outcomes after the second screening following Smith et al. (1999) and Williamson (2006). To do this, species invasion status had to be reclassified as either ‘invader’ or ‘non-invader’. We reclassified species twice, first with invaders representing spreading species only, and second with invaders representing both naturalised and spreading species. In addition we also calculated Cohen’s Kappa statistic for each invader/non-invader categorisation (Landis and Koch, 1977). The Kappa statistic can range from +1 (perfect agreement) to −1 (complete disagreement) with 0 indicating no agreement above that expected by chance.

We used receiver operating characteristic (ROC) statistics to assess the ability of initial H-WRA outcomes to differentiate between invaders and non-invaders, independently of the base-rate effect (Caley and Kuhnert, 2006; Gordon et al., 2008a). We also constructed ROC curves for naturalised species invading different habitats using overall H-WRA scores. Finally, to assess the relationship between H-WRA score, propagule pressure, time since introduction and invasiveness of naturalised species, we used a generalised linear model with a quasipoisson error distribution. We modelled the number of new compartments colonised per species as a function of H-WRA score, the number of plots planted (log transformed), and time since introduction (up to 2005), and selected the minimum adequate model using non-sequential backwards elimination and the likelihood ratio test.

3. Results

3.1. Summary of H-WRA data

On average, 38 questions (±0.23 S.E) were answered per species, with a minimum of 27 and a maximum of 46 questions. There was no significant correlation between H-WRA score and the number of questions answered \( (\rho = 0.05, P = 0.446) \), and there were no significant differences in the number of questions answered per species between invasion stages \( (F_{3, 204} = 0.985, P = 0.401; \text{Table 1}) \). Only nine questions were consistently answered for all 208 species. These were questions: 1.01 (is the species highly domesticated), 2.01 and 2.02 (climate match questions), 2.04 (native or naturalised in regions with tropical or subtropical climates), 4.01 (produces spines, thorns or burrs), 4.03 (parasitic – answered ‘no’ for all species), 4.08 (creates a fire hazard – answered ‘no’ by default), 5.01 (aquatic) and 5.02 (grass). A further 17 questions were answered for more than 90% of species. The questions answered for fewer than 25% of species were: 4.02 (is species allelopathic – 19%), 6.03 (hybridises naturally – 20%), 8.03 (well controlled by herbicides – 23%) and 8.05 (effective natural enemies present locally – 1%).

3.2. H-WRA scores and outcomes in relation to invasion status

The H-WRA score was significantly different, on average, for species of different invasion status \( (F_{3, 204} = 33.44, P < 0.001) \), with spreading >> naturalising > regenerating > surviving (Table 1). As was expected, rejection rates were lowest for surviving and highest for spreading species, and proportion of species requiring further evaluation was low across all categories (Table 1). However, two spreading species – the palm Arenga pinnata (Arecaceae) and the liana Landolphia owariensis (Apocynaceae) were accepted after the secondary screen (Table 1). Two other spreading species – Maesopsis eminii (Rhamnaceae) and Toona ciliata (Meliaceae) still required further evaluation after secondary screening.

The importance of correctly specifying whether a naturalised species is an invader or not, is illustrated by the impact on the reliability and accuracy of predictions. When naturalised species were classed as non-invasive, the probability that an accepted species will become invasive was only 1% and that a rejected species will not be invasive was 37% (Table 2). If naturalised species were considered invasive, the probability of an accepted species becoming an invader increased to 16%, and that a rejected species will not be invasive to 67% (Table 2). For both categorisations of invaders, the Kappa and ROC statistic suggested that the agreement between actual invasion status and H-WRA outcome was significantly
greater than chance and there was no significance difference in relation to how we grouped naturalised species (Table 2). The H-WRA outcome after second screening accepted 21 naturalised species, but rejected 20 species, and only left a small number of species in the ‘evaluate’ category (Table 1 and Supplementary material). Neither the number of plots planted (Wilcoxon rank sum test, \( W = 84.5, P = 0.806 \)) or compartments subsequently colonised (\( W = 118.5, P = 0.291 \)) by naturalised species differed significantly between those species rejected and accepted by the H-WRA. Across all species, there was a significant association between the decision to reject or accept and a record of invasiveness elsewhere (\( \chi^2 = 91.88, df = 1, P < 0.0001 \)), with 83% (43/52) of rejected species recorded as invasive elsewhere, compared to only 10% (14/139) of accepted species.

### 3.3. H-WRA scores and spread of naturalised species

When considering the number of compartments colonised by 47 naturalised and spreading species with available data in relation to number of plots planted, residence time and H-WRA score, the H-WRA score was the only significant variable retained after model selection. There was a significant positive relationship between the H-WRA score and the number of new compartments colonised (\( F_1,45 = 14.34, P < 0.001; 26\% \) deviance explained; Fig. 1). There were significant differences in average H-WRA score among habitats invaded by the 70 species that were spreading or naturalised (\( F_5,66 = 9.451, P < 0.001 \)), with species typical of open areas, forest edges and intact forest having significantly higher H-WRA scores, on average, than species restricted to plantations (Table 3). There were also significant differences between habitat groups in their environmental H-WRA score (\( F_5,66 = 6.418, P < 0.001 \)), again with forest, forest edge and open habitat species having significantly higher scores than plantation-only species (Table 3). In contrast, when considering the agricultural H-WRA scores, species typical of open disturbed habitats had significantly higher scores, on average, than forest, forest edge and plantation-restricted species (\( F_5,66 = 7.82, P < 0.001; \) Table 3). For overall H-WRA scores, the majority of intact forest species were rejected (Table 3), but two were accepted (\( A. pinnata \) and \( L. owariensis \)), and one open habitat species was accepted (\( Brugmansia suaveolens \)) whereas most species only naturalised in plantations were accepted (Table 3). The AUC for open habitat species was significantly greater than 0.5 but this was not the case for forest species, either at edges or for interior, closed canopy forest (Table 3). The AUC for species only naturalised in plantations was significantly less than 0.5 (Table 3), suggesting that the H-WRA was accurately identifying these species as having low invasion risk. The higher average H-WRA score and ROC sug-

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**Table 1**

Mean H-WRA scores (± standard error) for species of differing invasion status, and numbers of species (with percentages in parentheses) accepted, rejected and requiring further evaluation after H-WRA with second screen, and recorded as invasive elsewhere (‘yes’ to question 3.02, 3.03 or 3.04). Mean H-WRA scores with different letters are significantly different from one another (pairwise comparison \( t \)-test with Bonferroni correction).

<table>
<thead>
<tr>
<th>Status</th>
<th>Surviving</th>
<th>Regenerating</th>
<th>Naturalised</th>
<th>Spreading</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>H-WRA score</td>
<td>0.49 (0.46)(^a)</td>
<td>0.11 (0.80)(^a)</td>
<td>6.86 (1.00)(^b)</td>
<td>12.26 (1.39)(^c)</td>
<td>–</td>
</tr>
<tr>
<td>Questions answered</td>
<td>37.72 (0.33)</td>
<td>37.43 (0.51)</td>
<td>37.81 (0.44)</td>
<td>38.87 (0.79)</td>
<td>–</td>
</tr>
<tr>
<td>Record of invasion elsewhere (%)</td>
<td>18 (18)</td>
<td>6 (16)</td>
<td>22 (47)</td>
<td>17 (78)</td>
<td>–</td>
</tr>
</tbody>
</table>

**Table 2**

Numbers of species accepted, rejected and requiring further evaluation when categorised as invaders and non-invaders first with only spreading species as invaders, and second with naturalised species also classed as invaders (Su = surviving, R = regenerating, N = naturalised, Sp = spreading). Percentages in parentheses are the accuracies of correctly accepting non-invaders and correctly rejecting invaders. ROC curve (AUC ± standard error) and Cohen's Kappa statistics are also shown (CI = 95% confidence interval).

<table>
<thead>
<tr>
<th>Invader category</th>
<th>Non-invader (Su + R + N)</th>
<th>Invader (Sp)</th>
<th>Reliability</th>
<th>Non-invader (Su + R)</th>
<th>Invader (N + Sp)</th>
<th>Reliability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Outcome</td>
<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Accept</td>
<td>137 (74%)</td>
<td>2</td>
<td>0.01</td>
<td>117 (85%)</td>
<td>22</td>
<td>0.16</td>
</tr>
<tr>
<td>Evaluate</td>
<td>15</td>
<td>2</td>
<td>9</td>
<td>12</td>
<td>8</td>
<td>0.67</td>
</tr>
<tr>
<td>Reject</td>
<td>33</td>
<td>19 (83%)</td>
<td>0.37</td>
<td>12</td>
<td>40 (57%)</td>
<td>0.67</td>
</tr>
<tr>
<td>Total</td>
<td>185</td>
<td>23</td>
<td>70</td>
<td>138</td>
<td>70</td>
<td>208</td>
</tr>
<tr>
<td>AUC</td>
<td>0.91 ± 0.026; CI = 0.85–0.96; ( P &lt; 0.0001 )</td>
<td>0.79 ± 0.036; CI = 0.71–0.86; ( P &lt; 0.0001 )</td>
<td>0.576 (CI = 0.450–0.702)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kappa</td>
<td>0.432 (CI = 0.288–0.576)</td>
<td>0.576 (CI = 0.450–0.702)</td>
<td></td>
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**Fig. 1.** Number of compartments colonised by 47 spreading or naturalised species versus H-WRA score. Solid line = fitted model (26% of deviance explained, \( P < 0.001 \)); dashed lines = 95% confidence intervals of fitted model.
gest that H-WRA is better at discriminating invaders of open rather than forest habitats (Table 3).

3.4. Growth form

There were significant differences among growth forms in H-WRA scores ($F_{6, 198} = 15.00, P < 0.001$, Table 4), with herb, liana, and shrub/small tree species having significantly higher scores on average, than palms, trees and other phanerophytes (which included cycad, pandan, and other woody-stemmed monocot species). Herbs had the highest and palms had the lowest H-WRA scores. Lianas and bamboos were weighted more heavily in the H-WRA than other growth forms through questions 4.1 (smothering or climbing growth habit) and 5.02 (grass). However, whilst this appears reasonable for lianas, in ABG bamboos did not appear to pose a major risk. Removing this weighting reduces the average score of lianas and bamboos by one but this made little difference to the relative importance of these growth forms in ABG (Table 4). The majority of bamboos, palms, trees and other phanerophytes were accepted and the majority of lianas, herbs and shrubs/small trees were rejected by the H-WRA (Table 4).

4. Discussion

4.1. Comparing H-WRA performance with other tests

The problem of invasive plants represents a major force of global change, and their impact on biodiversity in the tropics is set to increase rapidly (Millennium Ecosystem Assessment, 2005). Efforts have been made to achieve an international standard for assessing the invasion risk of newly imported alien plant species (Panetta et al., 2001), and such an international standard can only be achieved by the H-WRA if it can identify high and low risk species with consistently high accuracy and reliability in all ecosystems at threat from invasion (Gordon et al., 2008a). The ability of the H-WRA to discriminate between invaders and non-invaders at this study site is comparable with previous H-WRA tests elsewhere (Gordon et al., 2008a), suggesting that the system works equally well at local landscape scales in the continental tropics. However, only if naturalised species are classed as non-invasive, does the ROC suggest the H-WRA performs as accurately as in other tests (Gordon et al., 2008a).

The reliability of correctly predicting an invader was also greater when only spreading species were classed as invaders. This is despite the low base rate of species classed as spreading (11% of all species). A lower base rate would be expected to decrease the reliability of correctly predicting an invader, i.e. increase the false positive rate (Williamson, 2006). However, reliability of correctly predicting an invader was lower when naturalised species were also classed as invaders, even though the base rate was higher (34%). This may indicate that our objective criteria describing naturalised species as those recruiting adults in fewer than 10 compartments are consistent with them being "non-invasive". Alternatively, the naturalised species category may be heterogeneous, with varying degrees of naturalisation among species. However, the number of compartments colonised did not differ between naturalised species that were accepted or rejected by the H-WRA. This is understandable since species distributions are dynamic and many naturalised species (particularly those rejected by the H-WRA) may in the future spread to further compartments and exhibit invasive behaviour. The significant relationship between the number of compartments colonised and the H-WRA score highlights that the screening protocol at least partially captures the risk of spatial spread. There are likely to be barriers to spread not encompassed by the H-WRA questions, such as micro-climatic effects on germination and growth, impacts of seed predators, and availability of efficient pollinators and seed dispersers (Richardson et al., 2000a,b). Propagule pressure and time since introduction were unable to explain spread of naturalised species in this study.

Table 3

Mean overall, environmental and agricultural H-WRA scores (±1 standard error) of species spreading or naturalised according to habitat colonised, and numbers of species (percentages in parentheses) accepted, rejected and requiring further evaluation. Outcomes after H-WRA are shown (overall). Mean scores for habitats with different letters are significantly different from one another (pairwise comparison $t$-test with Bonferroni correction). ROC curve statistics for overall H-WRA performance at identifying forest and open-habitat invaders are also provided—AUC = area under ROC curve ± standard error; CI = 95% confidence interval.

Table 4

Mean H-WRA scores (±1 standard error) of species according to growth form and numbers of species accepted; rejected and requiring further evaluation (after H-WRA second screen). The adjusted score does not include the values in the H-WRA protocol that apply specific weights to grasses and lianas. Mean scores for growth forms with different letters are significantly different from one another (pairwise comparison $t$-test with Bonferroni correction).
Despite the lower level of performance by H-WRA when naturalised species were classed as invaders, the system accepted the majority of surviving and regenerating species (non-invaders), rejected 83% of spreading species overall, and only 8% of all species required further evaluation after the second screening. The H-WRA cut-off score that maximised the difference between true and false positive rates was five or seven when naturalised species were classed as invasive, or as non-invasive, respectively, suggesting that the standard H-WRA threshold score of six is appropriate for this test site. Furthermore, Cohen's Kappa coefficient suggested that with a cut-off H-WRA score of six, agreement between H-WRA outcomes and invader/non-invader status was significantly greater than random.

4.2. Accepted invaders, growth form and habitat invaded

Whilst it is recognised that no predictive tool can identify all species likely to become invasive, the false negative rate (accepting invaders), should ideally be kept as low as possible (Panetta et al., 2001). Two spreading species were accepted by the H-WRA giving a false negative rate < 10%, which is comparable with other tests (Gordon et al., 2008a). Neither of these species is known to be very invasive elsewhere in the world, which emphasises both the weighting of this character in determining the final H-WRA outcome but also an inherent weakness in the protocol. There was a significant association between known invasion history and whether or not a species was accepted or rejected. Being a 'weed elsewhere' may largely explain why some surviving and regenerating species were rejected; 92% (11/12) of these species were known to be weeds elsewhere, compared to only 11% (14/126) of surviving and regenerating species that were accepted (Supplementary material).

In common with other tests of weed risk assessments (Scott and Panetta, 1993; Panetta et al., 2001), H-WRA was able to identify invaders of open habitats (analogous with 'agricultural weeds'), much more reliably than invaders of natural vegetation (i.e. undisturbed forest). The environmental H-WRA questions originally targeted at 'environmental weeds' (Pheloung et al., 1999) had a limited ability to identify forest invaders, despite the fact that no forest invaders were accepted after initial screening. This discrepancy may simply reflect greater complexity inherent in natural ecosystems and vegetation, thus making invaders of natural areas less easy to predict (Panetta et al., 2001; Rejmánek et al., 2005). As the original A-WRA was primarily developed for weeds of agriculture and more open habitats and not forest (Pheloung, 2001), it is perhaps unsurprising that forest invaders are rarely singled out as 'high risk' among naturalised species, in contrast to open-habitat invaders. It has been suggested that the rarity of natural area invaders (including forest invaders) compared to invaders of open habitats limits the capacity of H-WRA to predict 'environmental weeds' (Nishida et al., 2008). Daehler et al. (2004) found that H-WRA could identify invaders of natural ecosystems in Hawaii and the Pacific islands almost as well as invaders of human-altered ecosystems. However, natural communities and ecosystems of oceanic islands are rarely as undisturbed, diverse, or as structurally complex as continental equivalents, and are likely to be more easily invaded by alien species as a result (Denslow, 2003). Such species may be those invading open habitats, or a separate group with attributes associated with positive scores for agricultural as well as environmental questions in the H-WRA.

As might be expected, plant growth forms differed in their H-WRA score with herbs, lianas and shrubs identified as more invasive. Palms were consistently given low H-WRA scores compared to other growth forms even though several palms have spread in ABC. Low H-WRA scores and a subsequent greater likelihood of acceptance for palm species have been identified in Australia (Weber et al., 2009). Low H-WRA scores probably arise since palms are often single-stemmed and incapable of vegetative reproduction or of tolerating mutilation, have relatively large and few seeds per plant, and often do not reach reproductive maturity until after 4 years (Corner, 1966). However, palms are also often shade tolerant, dispersed by large birds and primates (Corner, 1966; Zona and Henderson, 1989), largely distributed within the tropics (Corner, 1966; Heywood, 1978), and resistant to herbivory (Grubb et al., 2008). Thus, the risk of palms becoming invasive in a continental tropical forest ecosystem may be greater than in the ecosystems and regions where the H-WRA was developed and tested. In contrast, shrubs and small trees may receive higher scores, as they may be more likely to form dense thickets, exhibit shade tolerance and tolerate mutilation compared to other growth forms. This study highlights that the H-WRA, originally developed in Australia and widely applied in temperate and sub-tropical regions remains an effective conservation tool for screening invasion threats to tropical ecosystems. However, the protocol is less effective at identifying risks to tropical forests than open habitats and does not adequately characterise the risk posed by palms to these ecosystems. One possibility would be to weigh question 4.09 “Is a shade tolerant plant at some stage of its life cycle” more heavily in the overall scoring at the expense of deleting question 4.08 “creates a fire hazard in natural ecosystems” to keep the maximum H-WRA score the same. However, it is not entirely clear at which stage of the life cycle, if any, shade tolerance is likely to facilitate invasion of forests. Modification of the shade tolerance question score would require support from sound experimental work testing the importance of shade tolerance in facilitating invasions into less disturbed vegetation such as forest. In addition, undesirable traits should be less focused on agricultural perspectives (e.g. palatability to grazing animals; host for pathogens and allergenicity) and more on impacts on biodiversity, hydrology and soil erosion. One aspect of WRA that still remains to be tested is the ability to distinguish high impact from low impact invaders; being invasive does not necessarily mean a species will have a high impact (and therefore cost) if introduced. Of the 12 questions in the WRA addressing potential impacts (4.01–4.12), only half were answered for all or most species. Thus, we suggest that a potential limitation of WRA could be that the system might be far better at predicting risk of spread rather than severity of impact. Indeed, Pheloung et al. (1999) do suggest that the WRA score reflects the probability that a species may become invasive, and not necessarily the degree of invasiveness.
Acknowledgements

This work is part of the Darwin initiative project “Combatting alien invasive plants threatening the East Usambara mountains in Tanzania” (162/13/033) and authors are grateful for financial support from Defra and NERC. The authors would also like to thank the staff of the Amani Nature Reserve, especially Mr. Corodius T. Save, and the Tropical Biological Association, especially Dr. Rosie Trewavely, for logistical support. Thanks also to Mr. Ahmed S. Mndolwa and Idydy Rajabu (Botanists, Tanzanian Forestry Research Institute), and Mr. Abduel B. Kajuru, for fieldwork assistance.

Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.biocon.2009.01.013. S1. Species assessed using the H-WRA, with scores and H-WRA outcomes. ‘Weed elsewhere’ represents answers to questions 3.02–3.04, where a ‘yes’ to any of the three questions means a species is a weed elsewhere.

References


Scots pine (Pinus sylvestris) trees on native vegetation in