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Abstract	Scientists, managers, and policy-makers need functional and effective metrics to improve our understanding and management of biological invasions. Such metrics would help to assess progress towards management goals, increase compatibility across administrative borders, and facilitate comparisons between invasions. Here we outline key characteristics of tree invasions (status, abundance, spatial extent, and impact), discuss how each of these characteristics changes through time, and examine potential metrics to describe and monitor them. We recommend quantifying tree invasions using six metrics: (a) current status in the region; (b) potential status; (c) the number of foci requiring management; (d) area of occupancy (AOO) (i.e. compressed canopy area or net infestation); (e) extent of occurrence (EOO) (i.e. range size or gross infestation); and (f) observations of current and potential impact. For each metric we discuss how they can be parameterised (e.g. we include a practical method for classifying the current stage of invasion for trees following Blackburn's unified framework for biological invasions); their potential management value (e.g. EOO provides an indication of the extent over which management is needed); and how they can be used in concert (e.g. combining AOO and EOO can provide insights into invasion dynamics; and we use potential status and threat together to develop a simple risk analysis tool). Based on these metrics, we propose a standardised template for reporting tree invasions that we hope will facilitate cross-species and inter-regional comparison. While we feel this represents a valuable step towards standardised reporting, there is an urgent need to develop more consistent metrics for impact and threat, and for many specific purposes additional metrics are still needed (e.g. detectability is required to assess the feasibility of eradication).

Keywords (separated by '-')	Biodiversity assessments - Biological invasions - Invasive alien species - Management - Impact - Distribution - Non-native
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Electronic supplementary material	Below is the link to the electronic supplementary material. Supplementary material 1 (DOC 51 kb)Supplementary material 2 (DOC 85 kb)

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ORIGINAL PAPER

A standardized set of metrics to assess and monitor tree invasions

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indication of the extent over which management is needed); and how they can be used in concert (e.g. combining AOO and EOO can provide insights into invasion dynamics; and we use potential status and threat together to develop a simple risk analysis tool). Based on these metrics, we propose a standardised template for reporting tree invasions that we hope will facilitate cross-species and inter-regional comparison. While we feel this represents a valuable step towards standardised reporting, there is an urgent need to develop more consistent metrics for impact and threat, and for many specific purposes additional metrics are still needed (e.g. detectability is required to assess the feasibility of eradication).

Keywords Biodiversity assessments · Biological invasions · Invasive alien species · Management · Impact · Distribution · Non-native

Introduction

The science of invasion biology has developed substantially (Gurevitch et al. 2011; Rejmánek 2011) but a recurring criticism of the discipline is the lack of an overall framework linking theory and management (Hulme 2003). Although several frameworks have been proposed to advance our understanding of invasions [e.g. Blackburn et al. (2011)], their development has largely been separate from schemes aimed at guiding management or policy (McGeoch et al. 2010; Rew et al. 2007). In contrast, conservation science has a well-established procedure for determining and reporting on the status of species—the IUCN Red Listing Protocol (Mace et al. 2008). Comparable listing efforts in invasion biology have largely focused on opinion (Lowe et al. 2000), but the need for a more quantitative approach is the same as for conservation science. There is an urgent need to move beyond basic lists of invasive taxa, to reporting information at a level that can be used to address various scientific and management needs (Fig. 1).

One of the major problems is that invasions do not follow administrative borders, so measuring the scale of a given invasion (and similarly the risk of extinction) often requires the integration of data collected by multiple stakeholders, agencies, and governments. While most countries are obliged to comply with international obligations (Box 1), data collection standards and the resources available for monitoring and control vary markedly around the world (Supplementary Material 1) (McGeoch et al. 2010; Nuñez and Pauchard 2010; Pyšek et al. 2008). Even within countries, different methodologies for quantifying invasions make it difficult to assess how invasions have changed over time (Guo 2011).

Any monitoring of an invasion also needs to be responsive over time-scales that are relevant for management. There is a real danger of responding unnecessarily to naturally variable populations or populations that ultimately fail to invade (Simberloff and Gibbons 2004; Zenni and Nuñez 2013). Nonetheless, responses need to be adaptive and rapid, particularly if eradication is to be a cost-effective option, and sustainable monitoring must have a clear outcome demonstrable in terms of specific agreed indicators. In comparison, for conservation assessments, population trends are measured over at least 10 years, whereas projections are typically framed over a century (Mace et al. 2008).

These issues can be addressed in part if a standardized global baseline for reporting biological invasions were to be developed. Such information needs to be relatively quick and inexpensive to measure or estimate, but should have the flexibility

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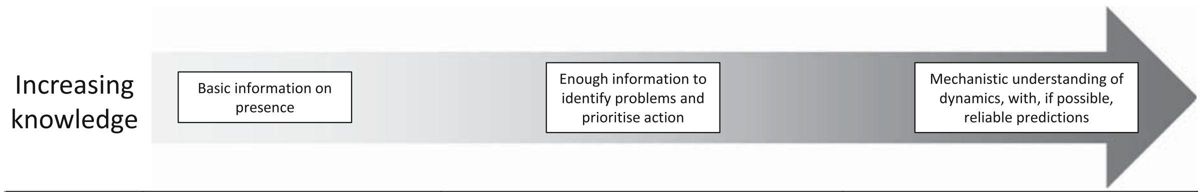

	Basic information on presence		Enough information to identify problems and prioritise action	Mechanistic understanding of dynamics, with, if possible, reliable predictions
Examples from conservation science	Lists of native species	Coarse-scale distribution data of natives	Native species ranked by conservation threat based on knowledge of population dynamics (i.e. Red List [1])	Detailed fisheries population models that incorporate multiple consequences under different scenarios of usage [2] optimal harvesting policies for forestry trees [3]
Examples from invasion science	Lists of alien species [4, 5]	List of alien species with status and basic information [6]	Knowledge of the status, abundance, spatial extent, and impact of alien species and how these characteristics are changing through time [this paper for trees].	Risk maps for probability of occurrence [7] Synthesizing different data to assess invasions across invasive spectrum [8] Prediction of risks of classical biological control releases [9]
Value of standardised metrics for invasion science	(i) use invasiveness elsewhere for risk assessments (ii) test for correlates of invasiveness		(i) facilitate cross-scale and between region comparisons (ii) meet policy obligations and guide policy development (iii) assess past and anticipate future trends and biosecurity risks (iv) global baseline (v) prioritise species or areas for management	(i) develop generic transferable management plans (ii) understand fundamental ecological processes (iii) compare evolutionary trajectories (iv) calculate costs and benefits of particular management approaches [10] (v) context-specific management of risk of impact

Fig. 1 Conceptual model of how increasing knowledge affects the potential for improving management and understanding, with examples from conservation sciences and invasion science. [1] Mace et al. (2008); [2] Worm et al. (2009); [3] Piazza (2010); [4] Richardson and Rejmánek (2011); [5] Lowe et al. (2000); [6] Pyšek et al. (2012); [7] Kaplan et al. (2014); [8] Ibáñez et al. (in press); [9] Martin and Paynter (2010); [10] van Wilgen and Richardson (2014)

to be built on in terms of complexity and utility so that impacts (and benefits) can be estimated (Fig. 1). Basic knowledge of whether a species is already present in the country and the current invasion status of its populations are important to determine what strategy and how much effort should be spent on management (Fig. 1). Additional information would facilitate fundamental comparative research in population dynamics, macroecology, and community ecology [work that is currently confounded by underlying differences in the way data on invasions were collected (Stohlgren et al. 2011)]. However, given invasions are context-specific, there is considerable value in deconstructing and evaluating the influence of species identity, dispersal potential, environment, and mode of introduction to develop a mechanistic understanding of the outcome of introductions (Fig. 1). Whatever the level of information available, if it is presented in standardized ways [or collected using common protocols (Gundale et al. 2014)], meta-analyses become powerful analytical tools to explore taxonomic and habitat differences (van Kleunen et al. 2010).

The aim of this paper is to recommend a standardized set of metrics to describe a tree invasion that will help assess progress towards specific management goals, and increase compatibility across administrative borders, and between invasions. We review metrics used to describe the presence of a species in a specified introduced range, though recognise that metrics at different levels (e.g. infra-specific, or at a community level) will provide important additional insights (Pereira et al. 2013). We focus on one specific group—introduced trees. Trees are relatively long-lived, individually identifiable, often are easily detected, can reach high adult densities, and, of course, are usually tall. Trees can therefore dominate plant communities and thus have a high potential to transform landscapes with profound impacts on biodiversity and ecosystems services (Richardson and Rejmánek 2011). Trees are an extremely polyphyletic assemblage of around 60,000–100,000 taxa (Petit and Hampe 2006), of which many species have been widely introduced beyond their native range. 434 introduced species (from <50 families) are invasive (i.e. ~0.5 % of total diversity) (Rejmánek and Richardson 2013), and more than half of these invaders have been introduced into several different biogeographic regions.

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Box 1 Challenges to developing lists of alien species for supporting monitoring and management

The listing of alien species is crucial for management and legislation, and many nations have committed to such listing in accordance both with relevant international conventions and national legislation. As signatories to the Convention on Biological Diversity (CBD), most countries are committed to mitigating national threats from alien species (including the enactment of relevant legislation) and reporting on the state of invasion in their countries. At the tenth meeting of the Convention on Biodiversity Conference of the Parties in Aichi, biodiversity targets were set for the period 2011-2020, with target 9 stating that "by 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment" (United Nations Environment Programme 2010). This commits nations to work towards identifying alien species present in their jurisdiction (Supplementary Material 1). The number of alien species in a country has been proposed as indicator to measure progress towards reaching the CBD 2010 Biodiversity Targets, specifically measuring the threat posed by invasion (McGeoch et al. 2010)

But how does one go about developing a comprehensive list of alien species for a given region? Not only is there limited expertise and available information but the development of lists of alien species is prone to numerous errors such as misidentifications, synonymies, insufficient surveys, impractical data resolution, lack of accessibility of data and insufficient information on native geographic distributions (McGeoch et al. 2012). To ensure consistent and comprehensive listing of alien species, the main sources of error need to be avoided (i.e. investment, consistency, transparency and standardization is required (McGeoch et al. 2012). Fundamental to listing alien species is the standardization of taxonomy (e.g. The Angiosperm Phylogeny Group for taxonomic placement, and www.theplantlist.org for accepted nomenclature) and terminology [e.g. see Pyšek et al. (2004) for standard definitions of biological invasion terms]. Regional context is an essential qualifier, particularly for large countries and nations where a species might be native in one part of the country but invasive in a different biogeographical area (Bean 2007)

A comprehensive list of alien species would require funding for exhaustive sampling and for sufficient expertise to facilitate identification. This has direct implications on management. Alien species that are most widespread and well known are likely to be recorded first. But in a country with an incomplete alien species inventory, naturalizing species not highlighted as problematic elsewhere are unlikely to be captured before they are widespread or damaging. The completeness of alien species lists varies between countries both as the amount of data available varies (i.e. the extent of local expertise and resources available to sample for and identify new species) and the number of species introduced varies (e.g. owing to differences in the size and sources of trade routes). The relatively short lists of aliens in developing countries are likely due to both effects (McGeoch et al. 2010). Such systematic biases hamper global comparative studies

Many archives have historically ignored alien taxa in collections (Fuentes et al. 2013; Zenni and Ziller 2011) and there is often inherent bias against collecting alien species. However, with the various sources of taxonomic uncertainty and changes to nomenclature, a physical record remains essential. Obtaining herbarium samples of flowering and fruiting trees can be logistically difficult (height and timing of flowering), but it is important for all alien taxa in a region to be catalogued. With changes in climate and nomenclature, and often substantial delays before the on-set of invasions, information on which trees are cultivated round the world is a vital background if the risks of future biological invasions are to be estimated

What characteristics of a tree invasion need to be included in a standardized set of metrics?

A standardized set of metrics for tree invasions has many possible advantages, but devising a list that would meet all requirements for all types of invasions is daunting [cf. McNaught et al. (2006)]. The metrics do, however, need to contain enough information such that they can be used to identify problems and prioritise action (cf. Red Lists in conservation science, Fig. 1). To achieve this, we considered that a set of metrics should provide information on status, abundance, spatial extent, and impact of an invasion and how these characteristics change through time. We argue that these characteristics of an invasion are necessary to: provide base-line statistics for biodiversity assessments; estimate impacts; estimate costs of different management strategies; estimate the threats posed; and ultimately place species into management and legislative categories as part of a strategic planning process. These characteristics are largely based on those used for conservation assessments (Mace et al. 2008), with the addition of a measure of impact. We reviewed published research on measuring each of these characteristics and proposed six representative metrics (Table 1).

Current and potential status

The most basic measurement of status in invasion biology is whether a taxon is present outside its native range (Pyšek et al. 2004). This is often the first information used for guiding biosecurity policy and management of alien invaders (Randall 2007). While such presence/absence lists are fraught with difficulties (see Box 1), invasive trees generally pose fewer

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Table 1 Key characteristics of an invasion, with the proposed standardized set of six basic metrics to allow for problem identification and the prioritisation of action (Fig. 1)

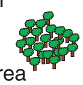
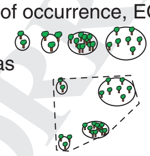
Characteristic	Recommended metric(s)	Uses of metric(s)	Additional metrics required for a more mechanistic understanding (Fig. 1)
Current status	(a) Category according to Blackburn's framework (not yet translocated, translocated, released into the wild, established self-sustaining populations, or invasive)	Placing species into management and legislative categories Providing headline statistics for biodiversity assessment reports	Status split into habitats, counties, protected areas, grid cells, biome and ecoregion. Genetic diversity. Residence time. Origin. Number, extent, and value of cultivated individuals
Potential status	(b) Potential range size from a species distribution model of climatic suitability 	Conducting a risk assessment Prioritising species for proactive management	Quantification of influence of barriers and mechanisms that could prevent a full invasion. Introduction-risk, species-based or area-based invasion debt quantified as appropriate, with estimate of how quickly it might be realized. Current and future pathways of introduction and dispersal identified and quantified
Abundance	(c) Number of invasion foci (populations) (d) Compressed canopy area (i.e. area of occupancy, AOO) 	Defining the number of foci requiring management Estimating management costs and current impacts	Number of individuals and stage/age structure of all invasion foci (populations), with information on reproductive output per individual. Size of seed-bank (if present)
Population growth rate	(c) + (d) change in abundance over time	-Planning control operations and determining management costs	Enough information to parameterize a suitable population growth rate model, e.g. a transition matrix, with some estimate of inter-annual and inter-site variation
Extent	(e) Area invaded (i.e. extent of occurrence, EOO) Either combined total if populations can be treated as separate OR alpha-hull ofall locations 	Estimating management costs and current impacts. Spatial planning of management efforts	Stage structure distribution of all individuals, seeds, and propagules. Define grain size (m or ha) and threshold level (% infested or presence/absence). Abundance and extent combined using scale-area curves
Spread	(e) change in extent over time	Spatial prioritisation of management efforts Conducting a risk assessment	A time-series of area invaded (ha) over time. A dispersal model that combines a landscape explicit natural seed (/propagule) dispersal kernel with routes of human-mediated transport. Both coupled to map detailing likelihood of recruitment

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Table 1 continued

Characteristic	Recommended metric(s)	Uses of metric(s)	Additional metrics required for a more mechanistic understanding (Fig. 1)
Impact	(f) Qualitative measure of likely impacts reviewed in the Australian Weeds Risk Assessment (AWRA) Protocol. An evaluation in terms of economics, cultural impacts, and biodiversity impacts	Placing species into management and legislative categories Providing headline statistics for biodiversity assessment reports Estimating current impacts	Costs and benefits (in economic and social terms) split up into different stakeholder groups and spatially explicit. Differences between invaded and non-invaded sites in terms of native species richness, abundances and evenness, changes in soil properties, increased production costs, loss of revenue owing to lower productivity
Threat	No specific metric proposed. A possible method is to identify whether a species might be a transformer or not (can use observations recorded in the AWRA), and whether the species is likely to over-top the recipient vegetation (Box 3)	Conducting a risk assessment Prioritising species for proactive management	Impact-based invasion debt quantified. Projections of how costs and benefits will change under different management scenarios with estimated costs and effectiveness to maintain current levels; to contain for a specified duration; or to eradicate. Global change scenarios considered, as well as potential for interactions with new introductions

Each proposed metric is denoted by a letter (a–f) as used in the main text

problems than other groups in this respect—trees are often intentionally introduced for use as ornamentals or for (agro)forestry, most can be easily detected, and native ranges are often well studied. However, we recommend that a recent herbarium specimen be set as the minimum required level of evidence for presence in a region (Box 1). Such lists presuppose the biological species concept, whereas invasions arguably happen at the gene level (Petit 2004). Therefore, some indication of sub-specific identity is valuable. While such information can often be gleaned from herbarium records, molecular analyses can provide important additional insights, pin-pointing areas of origin, reducing taxonomic misclassifications, identifying hybridization, and identifying differences between native and alien populations (Zenni et al. 2014).

Beyond presence and absence, other metrics for status are intrinsically composite, requiring information on abundance and spread. Standardized levels of information have been proposed for the nested dichotomies of non-introduced and introduced; non-naturalised and naturalised; and non-invasive and invasive species (Pyšek et al. 2004). Benchmark criteria (e.g. observed spread of more than 100 m within 50 years) has led to the development of a standardized invasive list for all trees and shrubs (Rejmánek and Richardson 2013; Richardson and Rejmánek 2011). Based on these criteria we developed a list of questions to determine the status of a tree introduction in a given region (Supplementary Material 2). The answers to these questions allow for species to be characterised following Blackburn et al. (2011)'s unified framework for biological invasions (the most recent and comprehensive such scheme) (Supplementary Material 2).

Predicting the potential status of a species at any particular place or time is problematic, though the basic criteria of invasiveness elsewhere and climatic suitability are good starting points (Hulme 2012). Estimates of potential status could also include an assessment of traits correlated to invasiveness and invisibility, and an assessment of the different mechanisms that might prevent an introduction becoming invasive (e.g. no suitable pollinator or dispersal agent). Developing a standard (and mathematically sound) metric for defining the probability that an invasion will result given particular conditions is a potentially valuable area of research (Leung et al. 2012).

One of the most basic limits to potential status is whether the region under consideration is climatically suitable or not. Species distribution models (SDMs) provide a good first estimate of potential distribution (Thuiller et al. 2005) and provide significant value for

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management, though the temptation to overstate the meaning of the quantitative results needs to be tempered by the various methodological and theoretical limitations to the approach (Guisan et al. in press; Nuñez and Medley 2011). Consequently, we recommend using a climate-based SDM as a first approximation of whether naturalization might be limited by physiology, but fine-scaled distribution predictions linked to probability of occurrence models are likely to be more useful for on-ground management (Brummer et al. 2013; Kaplan et al. 2014; Rew et al. 2006).

Abundance and population growth rate

At a broad scale, it is useful to estimate how many invasion foci there are, since the number of foci and their distribution have important implications for management. But while some invasions consist of distinct foci or populations, in many cases spatial distributions are more continuous. Based on reported tree pollen and seed dispersal distances (Petit and Hampe 2006), we suggest that foci separated by at least 10 km should have low levels of interaction and could safely be managed as distinct populations.

Conservation assessments, however, usually base abundance on the numbers of individuals and how they change through time. However, for tree species, individuals vary from seeds (which are small and numerous) to mature trees (which are large and much less numerous). As such there is a need to consider both numbers of individuals and the size and age (or ontogenetic) structure of populations. This is particularly relevant for species with large seed banks, where the size and longevity of seed-banks profoundly influence management decisions and outcomes (Panetta et al. 2011; Pieterse and Cairns 1988; Wilson et al. 2011). Size frequency histograms give some indication of likely population projections, but measurements of abundance, mortality, and fecundity over time are needed to calculate growth rate, while mechanistic and statistical models are needed to provide point estimates and rate predictions. Given the size and age structure of populations, invasive trees are particularly well suited to using matrix population models to estimate population growth rate [e.g. *Ardisia elliptica* (Koop and Horvitz 2005), *Gleditsia triacanthos* (Marco and Paez 2000), *Pinus nigra* (Buckley et al. 2005), *Prosopis* spp. (Pichancourt et al. 2012), and *Prunus serotina* (Sebert-Cuvillier et al. 2007)]. There are a variety of approaches for such models, but a projection matrix with 3 or 4 stages (seeds; seedlings and/or saplings; reproductive adults) and corresponding transition probabilities (incorporating survival, growth, and reproduction), is a reasonable minimum for many situations, allowing for estimates of finite rate of population increase (λ) or population (or metapopulation) growth rate [$r = \ln(\lambda)$] (Caswell 2001).

For some species, individuals can be hard to tell apart, and it is often difficult to count all individuals. Therefore, abundance is more readily estimated from the invaded area (i.e. condensed area or the net area of infestation) (Hui et al. 2009). This, in essence, is a measure of extent—area of occupancy (AOO) at a fine spatial scale—but as a simple metric of relative abundance for invasive populations it provides a useful link to impact and management. One method of calculating AOO is to assess the percentage of area covered, d, in an area of size, A. The condensed area (100 % equivalent cover) is simply $A \times d/100$. This provides a measure of local abundance, especially in canopy-forming tree species. This measure offers the benefit of being easy to calculate from gridded data and/or the use of digitally rectified aerial photography, without actually counting the number of individuals.

Extent and spread

Two measures have been adopted by the IUCN to describe the status of species' distributions (IUCN 2012) as they provide distinct, but equally valuable, information. First a raster-type approach can be used to describe the AOO for a particular unit (Gaston 2003) (e.g. quarter-degree grid or km^2 cells) giving an estimation of the abundance and the capacity to spread locally (and in this case we take it to be a measure of abundance rather than extent). Second, vector-type approaches, e.g. convex-hulls, can be used to circumscribe observations, where extent of occurrence (EOO) is that area which lies within the boundaries defined by the outermost geographic limits to the occurrence of a species (Gaston and Fuller 2009). We also note that surveys are never perfect. There are methods for describing uncertainty in distribution estimates due to imperfect detection (Mackenzie and Royle 2005), but a minimum requirement is to describe the area searched, when, and at what level of detail.

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If monitored through time AOO and EOO can be converted into area or distance over time to estimate spread rate (c), e.g. metres/year, km/year, or hectares/year. However, the appropriate units might depend on the spatial arrangement of spread (e.g. radial increase in an uniform area, or linearly along a watercourse), and both the rate and type of spread might change depending on the stage of invasion (e.g. initially slow spread, followed by exponentially increasing spread). Population models that account for dispersal are increasingly used to estimate spread (Caplat et al. 2012b; Smolik et al. 2010), but the data requirements can be daunting. Where possible, a plot of a time series of EOO measurements would be a good minimum but, again, for many situations data are not available. Trends in herbarium records over time (Aikio et al. 2010), and increases in the number of records obtained from surveys (Robertson et al. 2010) are useful for within-area measures, but interregional comparisons are challenging.

Specific methods for estimating spread include using a grid overlaid on aerial photographs and other remote sensing images such as high-resolution satellite imagery or radar data (e.g. Lidar). The occupancy of invasive trees can then be estimated, and a time series of images with the same grid location allows calculation of change in occupancy and extent metrics (Visser et al. 2014). Similarly, presence/absence transects can be repeated to obtain contingency tables including colonization and extinction rates. The colonization and extinction rates can be empirically modelled independently (Mackenzie and Royle 2005) or they can be fitted simultaneously along with the other two cases, cells remaining absent and cells remaining present (Jackson 2011), allowing estimates of spread rates. Simulations of this type allow managers to have a locally parameterized tool to test various management alternatives by simulating the effectiveness of different interventions over time and space (Caplat et al. 2014; Higgins et al. 2000). Several shortcut methods can also be useful for rapid estimation of spatio-temporal dynamics of invasive trees (Aslan et al. 2012). And an extremely useful aspect of trees is that various dating techniques (e.g. tree rings, morphometric measures, radio-carbon dating) can be used to age individuals in a population—historical extent can then be inferred from the spatial age structure allowing invasion reconstruction (Münzbergová et al. 2013; Richardson and Brown 1986). Finally, mechanistic approaches can be used, e.g. to predict seed movement across real landscape based on prevailing wind patterns (Caplat et al. 2012b).

Impacts and threats posed

The impact of an invasive species has been defined as the product of extent, average abundance, and effect per unit or individual (Parker et al. 1999). As discussed above, while measuring abundance and extent is reasonably straightforward, it is much more difficult to quantify the effect per individual or unit. Despite many useful conceptual models, a detailed quantification of impact is often precluded by data requirements, uncertainty, the non-linear nature of impacts, and the often complicated interactions between different types of impacts. Moreover, the negative effects of many invasions are likely underappreciated [poorly studied, difficult to detect, or due to a delay between invasion and impact (Simberloff 2011)], whereas positive effects are frequently overlooked and remain controversial. Given the difficulties of measuring impact, we recommend that relevant qualitative data should be collated and quantified whenever possible. One method for doing this is the Australian Weeds Risk Assessment (AWRA) protocol (Gordon et al. 2010). While many of the AWRA questions are not relevant to impact, and the AWRA was designed to be used pre-border, it is a useful and widely used standardised form. If the assessment is based on documented evidence it can provide a useful format for reviewing information relevant to impacts.

There could be substantial value in looking at how impact and threat are incorporated into risk assessments more systematically (Leung et al. 2012), and designing a scheme specifically for invasive trees. We propose that, for a baseline assessment for trees, two observations are used to determine threat—height in relation to native vegetation, and whether the species has a high risk of being a transformer [Box 2; Rejmánek et al. (2013)].

In short, the incorporation of standard metrics for impact and threat remains a major challenge. We believe that measures of effect per unit individual or area should be temporally and spatially explicit, and could be measured by cost (return or loss) on an area basis or for natural ecosystems by species extirpation per area over time. It would also be valuable to quantify how an introduced tree differs from co-

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Box 2 Categorizing invasion risk for trees

There are over 100 risk assessment models for invasive plant species (Leung et al. 2012), with some decision schemes developed specifically for trees or woody plants (Reichard and Hamilton 1997; Widrlechner et al. 2004). Any scheme investigating risk should, by definition, consider likelihoods and consequences. Here we discuss a simple way to allocate tree species to different categories of risk incorporating parts of the proposed standardized set of metrics

Likelihood of an invasion can be measured based on potential status on the invasion continuum and the likelihood of introduction or extent of planting. In the proposed set of metrics, climatic suitability is used as a coarse estimate of potential status, but this is in fact simply potential for naturalization. An estimation of potential status should also be informed by any a priori expectations that an invasion will occur, e.g. invasiveness elsewhere or the invasiveness of congeners. Invasiveness elsewhere is usually incorporated as a binary variable, but this is a true test of invasiveness only if the species has been introduced and had an opportunity to spread. Therefore invasiveness elsewhere can be expanded to include observations of the fate of introductions and the degree to which conditions where the known invasion occurred are similar to the conditions in the environment under consideration. More introductions to more regions, and a longer history and extent of planting should reduce the uncertainty as to whether a widespread invasion will occur (Wilson et al. 2011). A lack of invasions despite widespread planting forms the basis for proposed acceptable lists for horticulture (Dehnen-Schmutz 2011), and likewise repeated invasions in different biogeographic regions are indicative of a species that is highly likely to be invasive if introduced elsewhere. However, invasiveness elsewhere has little predictive power for those trees that have not been introduced or planted outside their native range (unless the original selection of species is correlated to invasive success, e.g. some types of forestry favour r-selected species). In the absence of information, the invasiveness of congeners can be used to estimate the *a prior* expectation of an invasion (Diez et al. 2012), as certain genera are over-represented in terms of invaders (Rejmánek and Richardson 2013)

Here we consider one component of the many consequences of an invasion, the potential threat to communities and ecosystems. We recommend two simple measures for trees—expected invader height relative to the expected canopy height of native vegetation (i.e. would the invader likely over-top native vegetation), and whether a species can be defined as a transformer. For the latter we use the nine categories of transformer as defined by Richardson et al. (2000)—excessive users of resources; donors/enhancers of limiting resources; fire promoters/suppressors; sand stabilizers; erosion promoters; colonizers of intertidal mudflats; litter accumulators; soil carbon storage modifiers; and salt accumulators. Transformer species have the potential to significantly affect ecosystem functioning and thereby services

The proposed analysis will not require much work in addition to the proposed metrics, as most pertinent information is included in the Australian Weeds Risk Assessment. But if the mechanisms underlying invasion and impact are understood, or if there are robust correlations with particular traits, then a more precise risk assessment, and more specific management recommendations, can be produced

Box 2 Figure 1 A proposed system for rapidly assessing the threat posed by an introduced tree. Darker shades indicate higher threat

			Consequence (in this case only negative consequences are considered, i.e. threat from Table 1)		
			Minimal	Medium	High
		Archetypal information	Many native analogues	Some key traits of a transformer species, or tall height	Traits of a transformer species, differs significantly in height and / or functional traits from species in threatened areas
Likelihood (Potential status x introduction risk)	very low	Widely planted for many years in multiple locations without naturalisation	Low Threat		
	medium	Some naturalisation occurs, and invasions under particular conditions.			
	very high	All introductions to physiologically suitable habitats result in an invasion			High Threat

occurring native species in key functional traits (e.g. water use, N-fixing, dominance) (Rundel et al. 2014), and estimate the benefits accrued against which any undesirable impacts can be evaluated (van Wilgen and Richardson 2014), though some components of impact and threat can be hard to quantify, e.g. the potential for hybridization with native species (Potts et al. 2003; Vanden-Broeck et al. 2012). The next step will be to develop networks of studies on impacts and, where possible, monitoring schemes should be modified to obtain information on the dynamics of the invader and the dynamics of the invaded community (both native dominants and species of concern).

Integrating metrics

There is substantial value in integrating these six metrics to improve our insight and management of invasive species. We discuss two possibilities here—current and potential status and impact and threat together provide insights for risk assessment (Box 2); and abundance, population growth rate, extent and spread are all related and if jointly considered will provide insights into invasion dynamics (Box 3).

A standard report

Using the recommendations above, we compiled information on a couple of notable invasions and present a standardised template for reporting tree invasions ("Appendix 1, 2"). Of notable interest is how the methods used to estimate the metrics vary, and how each carries particular levels of uncertainty.

Discussion

While lists of invasive species are extremely valuable (Rejmánek and Richardson 2013), indices are needed that can be used by decision makers and managers to estimate the state of invasions globally and how this will change through time (McGeoch et al. 2010). For invasive trees, we recommend as a minimum: (a) the current status of a species in a given region as defined by Blackburn's scheme (with regions ideally defined based on biogeography); (b) the potential status of the species (using modelling to estimate climate suitability); (c) the number of management foci (which should correspond to number of populations); (d) the condensed canopy cover (AOO at a very fine spatial scale); (e) the EOO for each management foci/population or the invasion as a whole; and (f) qualitative estimates of the impacts and threats posed (with information structured along the lines of the Australian Weed Risk Assessment Protocol). The methods for collecting these basic metrics are available although costly to obtain in some instances. More information will be required to answer specific question [e.g. estimates of the cost of eradication will require estimates of the detectability of individuals (Panetta et al. 2011)], and our proposal also does not include important aspects that are required for strategic planning [e.g. future population growth rates and spread rates (though a time series of AOO and EOO can be used to estimate past rates)].

There are several ways in which this set of six metrics could be expanded to incorporate other characteristics of an invasion, e.g. species-level traits, introduction dynamics, and traits of the recipient environment. There is an extensive and long-established literature on how intrinsic and extrinsic traits are correlated to the success of invasions and so can have value for risk assessments (Caplat et al. 2012a; Hui et al. 2011; Hui et al. 2014; Williamson and Fitter 1996). Species traits can also directly affect the utility of particular metrics. For example, for trees there is often very high seedling and sapling mortality but extended adult longevity, so simple measures of total numbers of individuals can be misleading both in terms of predicting population trends and for management. Seed bank longevity, age at maturity, generation time, and life span all provide important context and need to be estimated if the population dynamics are to be fully explored (Horvitz 2011; Petit and Hampe 2006; Rejmánek 2011).

Invasion dynamics are strongly influenced by the size, location, and number of introduction foci, i.e. the introduction dynamics (Wilson et al. 2009). The extent, spatial arrangement, and residence time of plantings will also affect the likelihood of an invasion being realized (Caplat et al. 2014). Moreover, if an invasion is realized, substantial conflicts can result between utilization and negative impacts affecting the management options available. As such, the history of introduction and current cultivated status provide important background information both for predicting the rate of an invasion, and for devising management strategies (van Wilgen et al. 2011).

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Box 3 Using the spatial structure of an invasion to provide management recommendations

Spread rate, abundance, and extent if considered jointly can provide important information for prioritising when, where, and how much management effort is required. They also provide vital information that can be used to classify invasive species. One approach for evaluating naturalized trees that included elements of spread rate, abundance, and extent was developed in Puerto Rico [1 = Slow spread and infrequent reproduction, 2 = Slow spread and abundant reproduction, 3 = Rapid spread and infrequent reproduction, 4 = Rapid spread and abundant reproduction; A = Abundant, C = Common, I = Infrequent or confined to limited habitats less than 100 hectares, R = Rare; (Francis and Liogier 1991)]. In outline it is similar to Rabinowitz's (1981) scheme for classifying different types of rarity. However, while both schemes provides useful approaches for thinking about and categorising invasions, they are less useful as management tools as the categories are binary and so the cut-offs are arbitrary and most species are likely close to the cut-off points. Moreover, at least for an extension of Rabinowitz's scheme, during the course of an invasion we expect species to change position, in part as a result of their introduction histories (Wilson et al. 2009; Wilson et al. 2007)

Box 3 Table 1 Classifying invasive tree species based on an adaptation of Rabinowitz's (1981) scheme for classifying rare species

		Extent of occurrence (EOO)[1]			
		Wide		Narrow	
	Habitat Specificity[2]	Broad	Restricted	Broad	Restricted
Fine-scale area of occupancy (AOO)[3]	Large	*Acacia dealbata* (Chile)	*Salix* spp. (Argentina)	*Hovenia dulcis* (Brazil)	*Melaleuca quinquenervia* (SE USA)
	Small	*Paraserianthes lophantha* (South Africa)	*Ficus carica* (California, USA)	*Araucaria araucana* (UK)	unlikely to be considered invasive

[1]Wide EOO would be >1 000 000km^2; or >50% of land area on an island; whereas narrow would be <100 000 km^2; or <10% of land area on an island (with 'average' distributions somewhere in between)

[2]A broad habitat specificity would be three or more vegetation types; whereas restricted would be confined to a single patchy soil type, e.g. serpentine soil in Europe, or a single vegetation type.

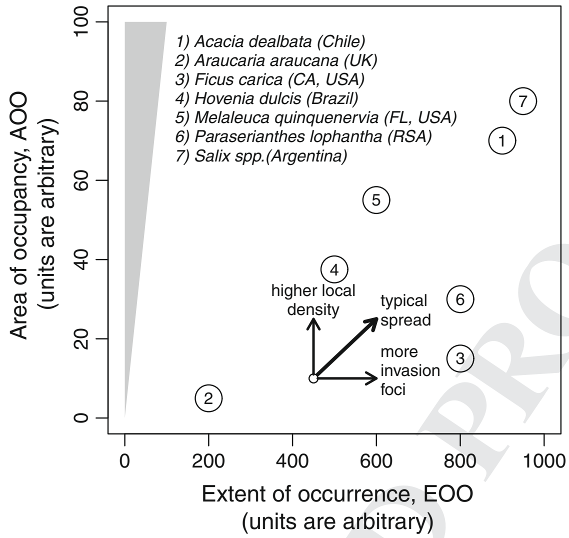
[3]The finescale area of occupancy is essentially a measure of population abundance for trees— either number of individuals per unit area or condensed canopy cover.

Another approach is to explicitly recognize that the patterns and processes underlying biological invasions change depending on the spatial scale investigated (Pauchard and Shea 2006). For example, scale-area curves have been used to estimate overall rates of growth and spread for species of conservation concern (Wilson et al. 2004), to determine the scale and trajectory of an invasion (Donaldson et al. 2014; Veldtman et al. 2010), and, in the context of native range dynamics, to predict invasiveness (Hui et al. 2011; Hui et al. 2014). Because of the complex nature of scale-area curves, a simple assessment of spatial pattern can be performed by combining area of occupancy (AOO) and extent of occurrence (EOO). The ratio of AOO to EOO gives a snapshot of the spatial aggregation of a species that is easy to calculate if gridded data of presence exists. Over time, an increase in AOO is likely to indicate an increase in canopy cover or abundance within a specific area, while an increase in EOO reflects range expansion. Managing a species that exhibits a temporal change in its distribution depends on whether both or one of the two measures is changing

We recognise that there are many further measures that could be added to an expanded list of metrics. However, it is important for managing invasions to have a mechanism that provides rapid assessments of the threat posed by an introduced species (Box 2). One could use a combination of key traits [e.g. the z-score proposed for conifers (Richardson and Rejmánek 2004)], together with an understanding of landscape features (e.g. habitat suitability; wind speed), and the nature of the introduction event [e.g. a lone tree as a point source vs. a plantation, fence-row or wind break (Zenni in press)]. We suspect that ensuring that the

Box 3 continued

Box 3 Figure 1 Plotting area of occupancy against extent of occurrence can provide useful insights into relative invasion dynamics. By definition AOO cannot be higher than EOO (grey area)



An invasion with a few large monocultural stands will have an AOO:EOO ratio close to 1, whereas a species with large extent but low occupancy (i.e. many small invasion foci) will have an AOO:EOO closer to 0. In these cases the first could represent a species with substantial local impact, but where containment to a few areas might be feasible, in the second case the species could be planted widely but has not spread much locally (e.g. a new popular ornamental introduction)

Trajectories in time can inform on the spatial dynamics of a species: spread by diffusion would result in a constant AOO:EOO ratio; while the formation of new invasion foci through long-distance dispersal would initially only increase EOO. If containment were successful, EOO should not increase, local clearing will initially reduce AOO, but EOO will only show a lasting decline if populations (including seed-banks) are extirpated

However, in some specific cases, scale-area curves or measurements of the AOO:EOO may underestimate invasions if there are no clear procedures to scale up or down. For instance, trees restricted to riparian corridors or strandlines will, by nature of the arrangement of suitable habitat, have constraints on their scale-area curves. Comparing range patterns between invasions is likely to be a substantial challenge and opportunity for invasion biology, and such patterns should be reported. For cross-scale management, see Caplat et al. (2014), and Kaplan et al. (2014)

metrics used to describe an invasion can be linked to traits and mechanisms will be a fruitful area of research, particularly when novel environments are likely to reshuffle existing communities and provide more opportunities for invasions to occur (Williams and Jackson 2007).

Conclusions

Tree invasions are causing important ecological and social impacts, but no consensus has been reached on how to measure and monitor them at regional and national scales. We hope this paper will stimulate discussion not just on how to quantify tree invasions, but also focus attention on determining the best and most practical variables and methods for estimating metrics, quantifying their uncertainty, and how these metrics should help guide policy and management. Our proposed set of metrics will facilitate this complex task, especially for invasions that cross administrative boundaries. These metrics provide the basis for assessing the success and failures of current management efforts and may help to improve future initiatives, particularly as it is expected that shifts in native species distributions in response to climate change will be analogous to invasions (Caplat et al. 2013). It remains to be seen whether each major functional or taxonomic group would need a new suite of metrics, but clearly extent is less easily measured for organisms that are more mobile as adults: inter-annual population fluctuations (and temporal invasion windows) might be important concepts that need to be captured. Whether a useful standardized set of metrics is achievable even for a single group like trees remains to be seen, but we feel that research in this area has the potential to advance the discipline as much as the processes for developing Red Lists has forced conservation science to develop a sound scientific base (Mace et al. 2008). The next step will be to trial the standardized set of metrics, revise the metrics in the light of practical experience, and develop practical guidelines for their measurement and reporting.

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Appendix 1: Example of species report (*Acacia paradoxa* DC. in South Africa)

Species: *Acacia paradoxa* DC. example herbarium record: (*Slater 7035*, BOL). No subspecific information available.

Location: South Africa.

Status: Invasive; D2 under Blackburn; **(in cultivation?):** not known to be cultivated recently (possibly introduced for ornamentation 100 years ago).

Potential: 6–13 % of South African land area; ~70–160 M ha (Zenni et al. 2009; Moore et al. 2011).

Abundance: ~12,000 plants (2010); 0.7 ha (condensed area); 70,000–700,000 seeds (2010).

Population Growth Rate: Few large individuals, 60–80 % of population <1 m and not reproductive in 2009; only 50 individuals >3 m.

Extent: 1 population; 350 ha (condensed polygon) 155–1,550 error rate.

Spread: natural radial increase of 100 m year^{-1} (assumed value), mostly gravity. Potential for seeds to be transported by road vehicles (not realized as yet).

Impact: Monoculture created; nuisance thorns. Impact ZAR 1,701 year^{-1} ha^{-1} (uncondensed area, 2,000 values) extrapolated from (de Wit et al. 2001). For an Australian Weed Risk Assessment see Zenni et al. (2009).

Threat: If potential area is multiplied by impact get to ZAR 100 billion year$^{-1.}$

Survey method(s) used: Systematic walked transects over ~700 ha to generate point distributions. At national scale distinctive species with road-side atlassing project (Southern African Plant Invaders Atlas), substantial on-going research into Australian acacias in South Africa by many field based researchers.

Notes: eradication plan in place.

Contact: alienplants@sanbi.org.za.

Information compiled by: John Wilson, jrwilson@sun.ac.za.

Refs:

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Appendix 2: Example of species report (*Pinus contorta* Loundon. in New Zealand)

Species: *Pinus contorta* Loudon.

Pinus contorta Loudon subsp. *contorta* = *Pinus contorta* Loudon var. *contorta.*

Pinus contorta Loudon var. *contorta.*

Pinus contorta subsp. *latifolia* = *Pinus contorta* var. *latifolia* Engelm. ex S.Watson.

Pinus contorta var. *latifolia* Engelm. ex S.Watson.

Location: New Zealand (numerous locations).

Status: Invasive; E under Blackburn; All four subspecies of lodgepole pine (*contorta*, *bolanderi*, *latifolia* and *murrayana*) have been planted (Miller and Ecroyd, 1987) and all regenerate naturally. (Ledgard 2001) **(in cultivation?):** Not known to be cultivated recently. Introduced in 1880 and established widely for erosion control during 1960s and 70s on a few thousand hectares and self-sustaining since then (Miller and Ecroyd 1987, Ledgard 2001). Suggested as possible covering ~100,000 ha by late 1990s (Ledgard 2001).

Potential: all already invasive.

Abundance: Various density stands. Seeds freely to high elevation and cones relatively young.

Population growth rate: Published information on estimated extent of cover (Miller and Ecroyd 1987, Ledgard 2001) suggests extent may be increasing at between 5 and 8 % per annum despite control efforts.

Extent: Numerous populations (many large and >1,000 hectares) totalling >100,000 ha extent at all densities. Many populations are found in remote locations as a legacy of where their establishment attempted to protect erosion-prone land from mass-movement. Due to their remoteness and potential cost there is little incentive address control or removal.

Spread: Natural radial increase of ~5,000 ha year^{-1} (assumed value), mostly wind and gravity.

Impact: Major visual transformation of iconic grazed grasslands into forest, with consequent recreational value loss and aesthetic impact. Invasions most problematic in low-stature native vegetation (Froude 2011), with up to 100 % loss of native plant biodiversity from high elevation grasslands (Ledgard & Paul 2008), strong shifts in fungal communities (Dickie et al. 2010) and, based on results from *Pinus nigra* strong effects on soil invertebrate diversity even at low tree-densities (Dickie et al. 2011). Economic loss through reduction in land for low-intensity grazing (sheep, beef-cattle). Loss of water a serious concern in some areas (Fahey & Jackson 1997).

Threat: Potentially 10–15 % of New Zealand land area; i.e. >2.5 M ha although could be greater. Highest threat is in conservation grasslands and alpine zone where removal will have high non-target impacts.

Survey method(s) used: No national objective survey or monitoring. One province (Canterbury Regional Council) has systematic estimates of extent of cover and density in 11 representative catchments ~70,000 ha to generate point and polygon distributions. Department of Conservation records the presence of weed species in a 10 × 10 km grid.

Notes: Limited control in a few locations.

Contact: Ian Dickie, DickieI@landcareresearch.co.nz.

Information compiled by: Larry Burrows, burrowsl@landcareresearch.co.nz.

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