Values and vulnerabilities: what are the values of the assets that are protected by Australia's biosecurity system and how vulnerable are they to incursions?

CEBRA Project 170713

Natalie Stoeckl[1], Aaron Dodd[2] and Tom Kompas[2]

[1] College of Business and Economics, The University of Tasmania
[2] School of BioSciences, The University of Melbourne

June 2021

Centre of Excellence for Biosecurity Risk Analysis

Acknowledgements

This report is a product of the Centre of Excellence for Biosecurity Risk Analysis (CEBRA). In preparing this report, the authors acknowledge the financial and other support provided by the Australian Department of Agriculture, Water and the Environment (DAWE), the New Zealand Ministry for Primary Industries (MPI) and the University of Melbourne (UoM).

The authors would also like to acknowledge their very many colleagues that have contributed to discussions, provided access to datasets, participated in workshops, reviewed content, and so on throughout this complex and ambitious project. In particular, the authors would like to acknowledge Paul Pheloung (DAWE), Ahmed Hafi (DAWE), Danny Spring (UoM), Karen Schneider (UoM), Callum Moggach (DAWE), Andrew Robinson (UoM), Richard Bradhurst (UoM), Sandra Parsons (DAWE) and Tony Arthur (DAWE); each of whom have made valued contributions to the project's final direction.

Table of Contents

Table of Tables

Table of Figures

Table of Appendices

Executive Summary

Invasive species pose a serious risk to a variety of industries, to people and to the environment. Identifying the magnitude of the potential harms caused by these hazards is generally accepted to be the first step in effectively managing them. However, most studies assess harm at relatively small [micro] scale (considering the impact of a single species, on a single industry or in a single region) complicating, or even preventing, the allocation of resources across risks and the many interventions used to manage them. In part, this reflects the difficulty of undertaking large-scale assessments, though, it is also a product of the different disciplinary norms within each of the environmental, agricultural and public health sectors. Thus, the aim of our project was to identify a framework for analysing impacts at the system [macro] scale – as biosecurity agencies need to do, in practice.

This aim contributes to our broader objective to estimate the net present value (NPV) of Australia's Biosecurity System (Dodd *et al.*, 2020). To the best of our knowledge, no one has ever successfully completed an economic evaluation of an entire biosecurity system and, during the first phase of this project, we identified several knowledge gaps preventing us from easily doing so (Dodd *et al.*, 2017). The most critical of these gaps was the lack of an overarching framework capable of appropriately characterising and aggregating the spectrum of harms arising in a multi-hazard, multi-intervention – therefore, multi-asset – context (though, see Bowen *et al.*, 2012; Liu *et al.*, 2014; Hafi *et al.*, 2015). We outline our thinking around such a framework here with a view to its use as the basis for our valuation (estimated via simulation) in the next phase of our project (see Dodd *et al.*, 2020).

We begin by summarising the available literature, clearly conceptualising the objective function, and articulating a generic framework that assesses hazards in the context of assets. Focusing on 56 terrestrial regions across continental Australia, we then undertake a large benefit transfer exercise to generate spatially explicit estimates of the current value of the ecosystem services (ES) and physical infrastructure (collectively, *assets*) that are protected by the biosecurity system (totalling ≈ A$250b p.a. across all regions and assets). Finally, we use this information to draw inferences about the relative 'vulnerability' of different assets (excluding agriculture) to 40 types of invasive species based on existing empirical estimates of impact and expert elicitation. Distinct heterogeneity was observed in both the spatial arrangement of asset values and their vulnerability to incursions.

In contrast to the existing pest-focussed biosecurity literature, our generic framework puts assets at the centre of the valuation; not only because considerably more is known about assets than hazards but also because the aim of any biosecurity system is to protect these assets from harm. Our regional estimates further allow us to consider asset vulnerability to a wide range of hazards and are thus of potential use in numerous policy settings, not just biosecurity. In particular, the spatial heterogeneities apparent in our estimates underscore the critical importance of spatio-temporal analysis for decision support – irrespective of the hazard. Though, most importantly, our framework standardises the kind of information required to estimate the harm caused by invasive species at the macro scale, potentially streamlining the collection of this information, by agencies, in the future.

1 Introduction

A key externality of the rapid increase in global trade and travel that has occurred since the beginning of the industrial revolution is the spread of biological organisms, both accidentally and deliberately, well beyond their natural biogeographic boundaries (Ricciardi, 2007; Hulme, 2009). While many of these species have contributed positively to their recipient region, a small proportion – those collectively referred to as 'invasive species' (*sensu lato*; including pests and diseases) – pose a significant risk to the economy, the environment and to public health (Williamson & Fitter, 1996; Pyšek *et al.*, 2012; Bradshaw *et al.*, 2016). Biosecurity is the practice of minimising the impacts of these species through a coordinated set of measures/interventions that aim to reduce both the likelihood that these introductions occur and the consequences that are incurred when they do (Nairn *et al.*, 1996; Beale *et al.*, 2008; Craik *et al.*, 2017). Thus, estimating the potential impact of a species, pest or disease, is generally accepted to be the first step in effectively managing them.

The valuation of impacts has two parts. One must first characterise the biophysical impacts that an organism has on its recipient community/host and then secondly assign a monetary value to those impacts (DeFries *et al.*, 2005; Boardman *et al.*, 2011). In agricultural contexts this is relatively straightforward because production losses and control costs can be directly measured, and prices of commodities can be observed in markets (Soliman *et al.*, 2010, 2015). Thus, most of the existing monetary estimates of the impact of invasive species relate to damages occurring to the agriculture and forestry sectors (Bradshaw *et al.*, 2016; Epanchin-Niell, 2017). In environmental contexts, however, this is more complicated as the biophysical processes are diverse, not easily observed, and many of them are not directly related to the market (Vilà *et al.*, 2010; Blackburn *et al.*, 2014). Consequently, the tools used in the so-called 'non-market' valuation literature are numerous, varied and complex (Bateman *et al.*, 2002; Getzner *et al.*, 2004; Farr *et al.*, 2016a; Hanley & Roberts, 2019).

Most commonly, impact estimates are developed to justify investment in a proposed management intervention, such as the application of import controls or the initiation of an eradication program. This has consequences for how the analyses are undertaken. For example, consequence estimates used to justify a trade restriction (e.g., Buetre *et al.*, 2006; Buetre *et al.*, 2013) must comply with the World Trade Organization Agreement on the Application of Sanitary and Phytosanitary Measures (the SPS Agreement; WTO, 1994) which restricts standing to producers. Conversely, a consequence estimate used to justify the eradication of a species with primarily environmental (including cultural) effects (e.g., Hafi *et al.*, 2013; Spring *et al.*, 2014) might focus solely on consumers. In either case, the implicit objective of the analysis is to determine whether the impacts exceed a threshold (e.g., that the Appropriate Level of Protection [ALOP] is exceeded, or that the program is cost-beneficial). Consequently, a common feature of these analyses is that they most frequently assess impacts at a small scale (*i.e.*, considering the impact of a single species, on a single industry or in a single region) because a more comprehensive analysis is unlikely to alter the decision (Dodd *et al.*, 2017).

Those charged with developing biosecurity policy, however, need information about the potential economic consequences of numerous biosecurity hazards that could individually or simultaneously impact numerous different 'assets' (be they economic, environmental, human or other) over time, and across space (Kompas *et al.*, 2019). Yet despite the fact that "single-species [hazard] oriented management philosophy has been increasingly replaced by an ecosystem approach" (Schlueter *et al.*, 2012), whole-of-system studies that simultaneously consider numerous impacts on numerous assets, belonging to numerous stakeholders, by numerous hazards are scarce and for the most part, qualitative/conceptual in nature (Liu *et al.*, 2014; Hafi *et al.*, 2015). Instead, many researchers approach large-assessment tasks by compiling estimates from multiple sources and aggregating (e.g., Bradshaw *et al.*, 2016; Paini *et al.*, 2016; Diagne *et al.*, 2021); although great care must be taken when doing so because compilation and aggregation issues are non-trivial and there are significant risks of double counting (Stoeckl *et al.*, 2014).

Consider the case where programs in the same region have had their impacts estimated using different valuation methods – for example, both the hedonic pricing and the travel cost method. Although both are perfectly valid micro-economic approaches that are well suited to the task of assessing the value of precisely defined changes to a small set of goods or services that generate impacts at small geographic scales, aggregating their results together (even though they are in the same units) could result in double counting (Holmes *et al.*, 2009; Boithias *et al.*, 2016). Recreational values appropriately estimated using the travel-cost method are known to be impacted by aesthetic values (regions of high aesthetic value attract more visitors). So, although the travel cost model focuses on recreational values, travel cost values will often have embedded within them, aesthetic values. Similarly, aesthetic values, most often estimated using hedonic pricing methods (e.g., comparing house prices in regions of differing aesthetic values), often have embedded within them, recreational values (houses in aesthetically pleasing areas likely also have good recreational opportunities). Therefore, to add both travel cost and hedonic estimates, generated for the same location, is to risk double counting. The small-scale nature of these analyses complicates the problem to such an extent that large-scale assessments all but demand a different type of approach.

Thus, the aim of our project was to identify a framework for analysing and comparing impacts at the system [macro] scale – as biosecurity agencies are ultimately required to do – to use as the basis for estimating the Value of Australia's Biosecurity System (Dodd *et al.*, 2020). Australia operates one of the most comprehensive biosecurity systems in the world, collectively spending in the order of A$1b on biosecurity activities annually (Craik *et al.*, 2017; Dodd *et al.*, 2017). Though, without a picture [framework] for piecing together the numerous individual estimates of the economic consequences of various incursions and biosecurity measures, it remains unclear what the magnitude of benefits generated from these investments are, because the situation is somewhat akin to having a handful of jigsaw pieces but no box – we're not sure of whether we have all of the pieces or even what the completed puzzle looks like.

The nationally endorsed definition of environmental biosecurity in Australia is as follows:

> *"Environmental biosecurity is the protection of the environment and/or social amenity from the risks and negative effects of pests and diseases entering, emerging, establishing or spreading in Australia".*

In this definition, 'the environment' includes Australia's natural terrestrial, inland water and marine ecosystems and their constituent parts, and its natural and physical resources. 'Social amenity' includes the social, economic and cultural aspects of the environment, including tourism, human infrastructure, cultural assets and national image (COAG, 2019). When assessing values and impacts relevant to such a biosecurity system it is thus clearly essential to consider a broad range of impacts on a broad range of what we hereafter refer to as *assets*.

Given the breadth of this scope, the Total Economic Value (TEV) framework (Pascual *et al.*, 2010) is an obvious potential choice for guiding our analysis (see Born *et al.*, 2005). Evolving gradually from (about) the 1940s, the TEV framework categorises benefits according to the way in which people benefit (i.e., derive utility) from environmental goods and services: *directly, indirectly,* or as a *non-use.* As one moves along the continuum from direct, to indirect, to non-use values, the link between benefits and markets becomes increasingly tenuous and the valuation task becomes increasingly complex. Although some researchers have attempted to argue that TEV can be estimated by adding direct-use, indirect-use and non-use values, the values cannot be assumed to enter the utility function in an additively separable manner (see Carbone and Smith (2013) who highlight this problem when discussing the difficulty of trying to use non-market valuation estimates, derived from partial equilibrium methods, in a computable general equilibrium model). Consequently, adding values using a TEV framework risks double counting. This is not a critique of the TEV framework: it was not developed with the intention of being used to guide large-scale 'whole of system' valuation exercises, but rather to highlight the diversity of values associated with the environment.

As part of the development of the Risk-Return Resource Allocation (RRRA) Model (described in Craik *et al.*, 2017), Bowen *et al.* (2012) briefly reviewed non-market valuation approaches, with a view towards considering ways of including non-market values in macro scale biosecurity assessments. Their general conclusion was that it would likely be too costly to conduct separate non-market valuation studies for each potential impact (suggesting the use of general scales instead), but – most pertinent here – they highlight the need for using a framework for thinking about impacts. Their work is extended in Chesson *et al.* (2014), who look at biosecurity impacts on several different 'capitals', using various ecosystem services (ES) as a type of natural capital. Like the TEV, however, the RRRA capitals framework was not developed with the intent that it be used as a framework for organising 'values' for whole-of system valuation, and its use could lead to double-counting.

Derived from the original Millennium Ecosystem Assessment classification system (MEA, 2005), the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2012) is another framework for thinking about various ecosystem services in a systematic manner that minimises potential problems of double counting related (ecosystem) values. It includes all the different types of direct-use, indirect-use, option and non-use values identified in the TEV, but refers to them using different terminology, and groups them in different clusters. If done carefully, one can use this framework to guide the compilation of information about the value of ES, to generate a final composite estimate of the value of all ES from a particular area or region without double counting (see Costanza *et al.*, 1997). The framework does, however, exclude many potentially important biosecurity-related impacts currently captured within the RRRA framework (specifically those unrelated to the environment), and could not therefore, be used without adaption.

We begin by summarising the available literature, clearly conceptualising the objective function, and articulating a generic framework that assesses hazards in the context of assets. Focusing on 56 terrestrial regions across continental Australia, we then undertake a large benefit transfer exercise to generate spatially explicit estimates of the current value of the ecosystem services (ES) and physical infrastructure (collectively, *assets*) that are protected by the biosecurity system. Finally, we use this information to draw inferences about the relative 'vulnerability' of different assets to 40 types of invasive species based on existing empirical estimates of impact and expert elicitation.

Note: our value estimates are not standard measures of welfare/wellbeing. Data deficiencies make it impossible to use such standard measures. We detail the measures used in each section.

2 Methods

We begin this section by conceptualising the process of valuation (section 2.1). This clarifies that one can 'value' the biosecurity system in different ways: compiling estimates of biosecurity values directly from the literature; or, first estimating the value of assets 'at risk', and second determining what proportion of the 'at risk' assets could be damaged by incursions. We then review relevant literature (section 2.2), finding that data deficiencies preclude option one, our task thus being to first determine the value of assets that are protected by Australia's biosecurity system. In section 2.3 we give an overview of methods used to value each asset, with additional methodological and data details provided in the appendices. Section 2.4 describes our approach for estimating the relative vulnerability of assets to different types of invasive species, completing our framework for valuation.

2.1 Conceptualising the process of valuation

We conceptualise the problem of trying to 'value' the biosecurity system (or indeed any biosecurity measure) in Table 1 – noting the importance of the counterfactual. This conceptualisation greatly simplifies reality, but serves a useful purpose, demonstrating that one can in principle estimate the value of a biosecurity measure in two different ways:

1. Estimating (or collating existing estimates of) the damages likely to occur to assets in the absence of a biosecurity measure (≈ column *III*, Table 1), or
2. Estimating (or collating existing estimates of) both:
    a. existing values, with the current measures in place (≈ column *I*, Table 1); and
    b. expected values in the absence of a biosecurity measure (≈ column *II*, Table 1)

    and then subtracting (II) from (I).

Table 1: Conceptualising the whole-of system valuation problem. Observed values are highlighted in grey emphasising the importance of the unobserved counterfactual when estimating damages.

Good/Service (asset)	**Observed values with existing biosecurity measure I**	**Expected values in the absence of biosecurity measure II**	**Estimated (expected) value of biosecurity measure III**
Benefits			
Market goods	A_1	A_2	$(A_1 - A_2)$
Non-market goods	B_1	B_2	$(B_1 - B_2)$
Costs			
Control costs	C_1	C_2	$(C_1 - C_2)$
System costs (expenditure)	S	0	S
Total	$(A_1 + B_1) - (C_1 + S)$	$(A_2 + B_2) - (C_2 + 0)$	$(A_1 - A_2) + (B_1 - B_2) - (C_1 - C_2 + S)$

What can be done in practice, however, will depend upon resources available and upon the presence/absence of studies that provide data about 'damages' which can, like 'benefits', be transferred. Before deciding how best to approach the problem, we thus reviewed literature to assess the extent of knowledge.

2.2 Knowledge state

We sought to identify the knowledge state by collating published papers relating to the economics of biosecurity and categorising the subset generating monetary cost/benefit values according to the broad type of hazard evaluated, the type of asset considered, and the type of value generated. Our overarching aim being to determine which assets/hazards we have the most/least knowledge for. As an initial starting point, we chose to work with broad taxonomic categories to prevent sparsity given the large number of potential asset x hazard combinations and known taxonomic biases (Pyšek *et al.*, 2008). Drawing on insights from other researchers who have worked at similar scales (e.g., Akter & Grafton, 2010; De Lange & van Wilgen, 2010), we adopted the following categories:

1. Terrestrial plants;
2. Terrestrial invertebrates;
3. Terrestrial vertebrates;
4. Pathogens (n.b., plant and animal pathogens were considered separately in later studies);
5. Freshwater species (all); and
6. Marine species (all).

For assets, we used categories that align with the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2012) – albeit with minor variations to adequately cover all assets of concern to Australia's biosecurity system, and to ensure that categories are adapted for context (after Díaz *et al.*, 2018). Provisioning services were nominally grouped according to whether they fall within scope of the Australian Government's Agriculture *Portfolio Budget Statement*, or not. We also explicitly noted that amenity values are a type of cultural service, and added the additional categories (companion animals, physical infrastructure, human and social capital) identified by Chesson *et al.* (2014) for use within the RRRA model. Our asset framework, grouped into broad categories, is shown in Table 2. A detailed description is included in Table 8 (Supplementary Tables).

Table 2: Asset classes used in the initial analysis.

Relevant Capital	Asset Type	Asset Class	Sub-class
Natural	Provisioning	Portfolio Industries	Agriculture
			Forestry
		Non-Portfolio Services	Indigenous Subsistence
			Water for Consumption
	Regulating	Erosion Control	Erosion Control
		Flood Control	Flood Control
		Genepool / Nursery	Genepool
		Carbon Sequestration	Carbon Sequestration
		Mediation of Soil / Air	Toxin Mediation
	Cultural	Residents – Use	Recreation / Aesthetics
		Residents – Non-Use	Existence / Bequest
		Non-Residents – Use	Tourism
		Indigenous – Non-Use	Indigenous
	Companion Animals	Pets (Cats, Dogs, etc)	Domestic Animals
		Horses (non-racing)	Recreational Horses
Physical	Infrastructure	Dwellings / Utilities	Infrastructure
Human	Human Health	Human Health	Human Health
Social	Social Infrastructure	Social Infrastructure	Social Infrastructure

We began by updating the dataset compiled by Dodd *et al.* (2017) to include additional recent publications relevant to our analysis; this compilation numbered 268 articles. More than one half of these studies did not report monetary estimates of damages, control or eradication costs in a way that could usefully inform an empirical assessment, so we focused on the 117 studies that did. We then counted the number that provided empirically 'transferable' estimates of the potential (or actual) damages inflicted by each broad hazard type on each asset category, and on control or eradication costs (Table 3). Some studies provided cost/benefit estimates for more than one asset or hazard. Some studies did not assess costs/benefits by asset category, instead considering control or eradication costs associated with a hazard; and some studies provided separate estimates of control/eradication costs and costs/benefits associated with particular asset types. Consequently, the total number of estimates identified in Table 3 (262 internationally) exceeds the number of studies (117). The total number of estimates was 262, a small subset of which (52) were Australian.

Our compilation of studies is neither definitive nor exhaustive, so does not describe the entire body of literature and, therefore, under-states the true availability of information[1]. Nevertheless, it demonstrates that most empirical work is concentrated on a subset of hazards (terrestrial), assets (agriculture and forestry) and cost categories (control costs). Internationally, and in Australia, more than 50% of estimates relate to control costs or yield losses (Table 3). About 16-17% of studies provide estimates relating to the potential damage that invasive species could cause to regulating services. Other research is sparsely scattered across the remaining asset categories. Most notable are the knowledge gaps relating to the monetary 'value' of incursions on the 'social' or 'companion animals' asset categories (there were several studies focusing on the Hendra virus, and its impact on horses, and several talking about the social impacts of various hazards, but none were monetised). Evidently, there is enough knowledge to draw inferences about the impact of some hazards on some assets, but there is insufficient knowledge for a comprehensive, whole-of-system assessment.

That said, Table 1 clearly shows that compiling estimates of 'impact' (akin to collating studies relating to column III) is not the only way to approach the problem: it is possible to instead, gather data relevant to columns I and II and compare. Noteworthy here is the fact that studies which consider non-market asset values in general (e.g. column I) are much more prevalent than those that only consider asset values in a biosecurity context (Holmes *et al.*, 2009). The Economics of Ecosystems and Biodiversity (TEEB) valuation database (van der Ploeg & De Groot, 2010), for example, contains no fewer than 1310 value estimates, 116 of which are Australian. We thus chose to approach the whole-of-system valuation problem by determining first, the value of assets at risk (section 2.3), leaving open the task of predicting the impact of hazards on 'at risk' assets (i.e., their vulnerability) for later analysis (section 2.4).

---

[1] A more targeted search of studies within the field of health economics would likely identify many more empirical studies that assess the impact of pathogens on human health, but more targeted searches are unlikely to uncover a substantive body of empirical research relevant to other assets. In addition, there is likely to be at least some double counting in this summary, because the studies using benefit transfer (BT), may be referencing other studies that are included in the table.

Table 3: Count of empirical studies focused on the economics of biosecurity by asset and hazard.

Total numbers shown in black; numbers relating to Australian studies and Australia studies that assessed 'consequence' shown in grey; asset/hazard categories for which we could find no potentially transferrable estimates shaded in grey. Numbers relating to 'consequence' indicative only.

	Hazard category								
Asset Category	Terrestrial Invertebrates	Terrestrial Vertebrates	Terrestrial Plants	Terrestrial Pathogens	Freshwater (all)	Marine (all)	Aquatic (all)	Not specified	Total
Portfolio	18	5	11	19	4	2		5	64
Australian	4	1		5				2	12
Consequence	2							1	7
Non-Portfolio	1	1	3	1			1		7
Australian			1	1					2
Consequence									
Regulating	1	3	12	1	1	1	1	3	23
Australian		1	1	1				1	4
Consequence									
Cultural	10	5	13	3	4	5	1	3	44
Australian	3	1	2	1				1	8
Consequence	2		1						3
Infrastructure	6	3	4	1	3				17
Australian	1			1					2
Consequence									
Domestic Animals									
Australian									
Consequence									
Health	2	2	2	2	1			1	10
Australian	1							1	2
Consequence									
Social									
Australian									
Consequence									
Not specified	5	3	1	1	0		2	2	14
Australian	1	1	1						3
Consequence			1						1
Control costs	18	10	13	16	4	2	3	4	70
Australian	3	3	2	3			1	1	13
Consequence	1		1	1					4
Eradication &/or exclusion costs	4	4	1	2			1	1	13
Australian	1	2		2			1		6
Exclusion				2			1		3
TOTAL	65	36	60	46	17	10	9	19	262
Australian	14	9	7	14			2	6	52
Consequence/exclusion	5		3	7			1	2	18

2.3 Estimating asset values ‘at risk’

Although it is possible to simulate the spread of a species at fine geographic scale, the scale at which it is possible to estimate and thus spatially allocate asset values depends, amongst other things, on the geographic scale at which relevant economic data are available. Generally, the smaller the geographic area considered, the less available are economic data. Most of the data required to inform an assessment of the market and non-market value of Australia’s biosecurity system are available for Natural Resource Management (NRM) regions (DoEE, 2017) – and it is on these regions that we focus. Figure 1 shows the boundaries of these regions with population densities, clearly highlighting the concentration of people around the coastline – particularly around urban centres.

We chose to exclude the marine environment and to also exclude human and social capital when assessing values. These decisions were driven by a desire to develop ‘defensible’ estimates. Our background investigations highlighted that there was insufficient economic data (or underlying knowledge) to accurately estimate values associated with human and social capital at the NRM scale or to assess values in the marine environment (where geographic boundaries are transcended by both socio-economic and natural systems/interchanges). To have included the marine environment or social and human capital would, we feel, have required us to make indefensible assumptions which could risk eroding confidence in other estimates.

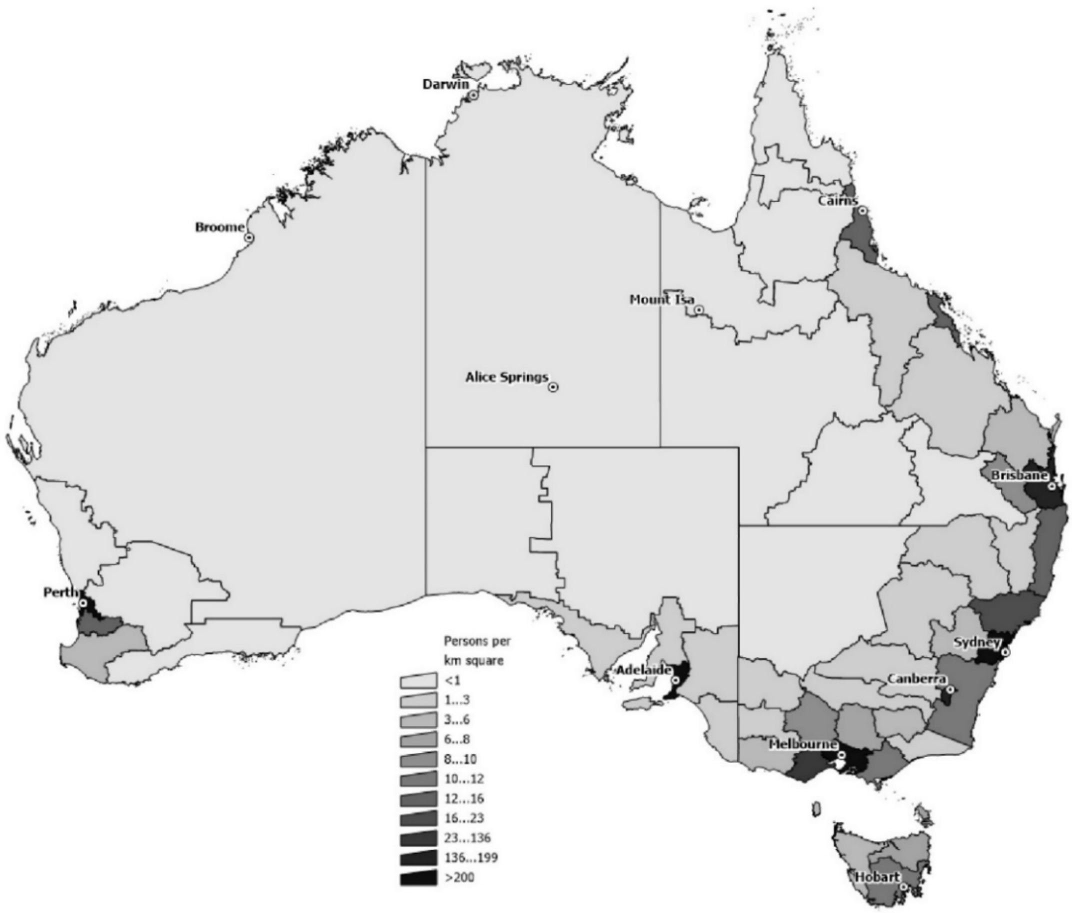


Figure 1: The geographic resolution of our analysis: Natural Resource Management (NRM) regions and their population density.

2.3.1 Provisioning Services I: 'Portfolio' Industries

Agriculture

The Australian Bureau of Statistics (ABS) provides data about the value of agricultural production, at NRM scale, for broad commodity groups (ABS, 2018). We used that data, in conjunction with land-use data, to generate estimates relevant to portfolio assets. Although data relating to the value of production is available for relatively fine-scale commodity groups, we work with more aggregated data that can be linked to the two-digit land-use classification of the Australian Land Use Mapping (ALUM). Estimates thus relate to cropping, horticulture and livestock (intensive (e.g. pigs, poultry), and extensive (e.g., sheep, cattle)). For each commodity group, we divided the ABS estimates of production values, by the hectares of land used for that type of production, to estimate NRM specific (per hectare) values. Variations in per-hectare values reflect differences in products mixes (e.g. hay, sugar) and in the economic, social, meteorological and biophysical conditions which create differences in costs and productivity across NRMs. This captures heterogeneity across NRMs though data deficiencies prevent us from identifying intra-NRM heterogeneity – which could be substantial in the larger NRMs such as the Northern Territory (NT) and Rangelands in Western Australia (WA).

Forestry

Forests account for more than 16% of Australia's land area (ABARES, 2017), although not all forests are logged – so only a portion of forested areas provide a provisioning service (all provide regulatory services, discussed later). Data relating to the value of logs produced are not available at NRM level, so we combined Australia-wide value-of-production data (ABARES, 2017), with ALUM data to estimate the average (Australian) value per hectare for forest activities. Estimates of the Australian average value per hectare of forest activities were then linked to land use classifications to infer whole-of NRM estimates. We note that more sophisticated methods of estimating forestry values are multi-period and take into account both growth and harvest rates – the aim being to estimate net present values (see Creedy & Wurzbacher, 2001). While much more simplistic than these approaches, our data-constrained approach of simply calculating average values per hectare from production data is consistent with our overall expected value approach outlined in Table 1.

2.3.2 Provisioning Services II: Non-Portfolio Services

We consider both Indigenous subsistence values and water values as non-portfolio provisioning services. However, to estimate water values we use a supply-side approach that is inextricably linked to the methods used to estimate other related regulatory service values, so although (drinking) water values are strictly speaking, classified as provisioning services under the CICES, we postpone their discussion until section 2.3.3, explaining related methods together.

Indigenous Subsistence

Sangha *et al.* (2019) used insights from the literature to estimate Indigenous subsistence food and material values for the Northern Territory: ≈ $500 per (Indigenous) person. This estimate is relevant to Indigenous people who live 'on country' but may be somewhat less relevant for Indigenous people living in large cities (when access to 'country' is more difficult). To allow for that:

1. We calculate an 'area weighted' ARIA+ score for each NRM using the 2011 ARIA+ (Accessibility/Remoteness Index of Australia) scores for each 2011 ABS defined SA1 region within the NRM: NRMAria+ = $\sum_{i-1}^{N} \frac{Areas_i \times ARIA+_i}{Area_{NRM}}$
2. We estimated (per Indigenous person) subsistence food values within each NRM as a proportion of maximum values ($500) – that proportion calculated as: $\frac{\text{NRMAria+}}{15}$. The most remote regions (with NRMAria+ scores of 15) thus have food values of $500; in urban areas (with NRMAria+ scores of 0), subsistence food values are zero.
3. All-of-NRM Indigenous subsistence values were estimated by multiplying these Aria-weighted per-person values by estimates of the Indigenous population. This assumes that the value per-person is constant throughout each NRM – though, it is possible to derive estimates that are more spatially disaggregated by, for example, multiplying per-person values by small area population estimates, to derive small area values and re-aggregating.

We consider other Indigenous cultural values (albeit inadequately) – in section 2.3.4.

2.3.3 Regulating Services (and Water)

First, we consulted van der Ploeg and De Groot (2010)'s database, identifying studies that had generated estimates of water and regulating service values. We used sub-categories of ES from the CICES (Haines-Young & Potschin, 2012) to guide the compilation of data and minimise the risk of double-counting. We focused on: water purification (including, but not limited to water for human consumption); erosion control; flood control; gene-pool/nursery values; carbon-sequestration; and the mediation of soil and air. We also conducted additional literature searches to identify more recent estimates, focusing primarily, although not exclusively on studies undertaken in Australia, or in arid regions (which comprise so much of the Australian landscape).

Second, we developed a concordance system to match the descriptors of ecosystems used in the ES value database (hereafter *biome*) to the Major Vegetation Groups (MVGs) used in the (Australian) National Vegetation Information System (NVIS; DoEE, 2018), so that we could allocate values spatially (Appendix A). We then used information from each study in our compilation of data, to determine which 'biome' each (sub) ES value estimate referred to.

Third, we estimated transfer functions for each (sub) ES/biome. If seeking to map ecosystem service values at regional scale, one must consider a wide range of socio-economic factors such as the spatial pattern of population distribution (Paracchini *et al.*, 2014) in addition to supply-side factors such as type, quantity and 'quality' of biome (Tardieu & Tuffery, 2019). It is also important to consider the availability of substitute sites (Tardieu & Tuffery, 2019) and income – since the income-elasticity of demand for environmental goods and services is not constant (Barbier *et al.*, 2017)[2]. We, therefore, sought to reduce transfer errors by using (benefit) transfer functions.

There is much controversy about the best transfer approach and about the best variables to use within transfer functions (Baker & Ruting, 2014), with evidence to suggest that one can reduce transfer errors by ensuring that transfer functions include socio-economic (Johnston *et al.*, 2017), geospatial (Fitzpatrick *et al.*, 2017) and 'perception' variables (Farr *et al.*, 2016b), all of which have been found to influence value estimates. We reviewed several meta-analyses of studies undertaken

[2] Values will depend not only on biophysical, but also upon socioeconomic context. For example, aesthetic, amenity and recreational values describe the benefit that people derive from being able to enjoy an area's aesthetic, amenity and/or recreational services. If there are no people to recreate or to appreciate these things, then by construct, their actualised value must be zero.

across multiple countries to identify socio-economic factors which are commonly found to be statistically significant correlates of ES values (Appendix B). Noting there is a difference between what should be done and what could be done, we focus on three core variables since they were readily available for most study and transfer sites. Appendix C provides data by NRM (our transfer sites). For study sites, we compiled information on size of the study site, population of country, area of country, and GDP per capita (for the year in which the study had taken place). Abundance was estimated as area of study site divided by area of county. If there was insufficient information provided in the primary study, we sought such information from FAO (2019), The World Bank (2019).

We converted value estimates from our compilation to 2015 AUD (per hectare, per annum) and used that data (N=208) to estimate transfer functions. This was done by regressing value estimates against estimates of biome abundance (area of study site/area of county), population density and GDP per capita. We sought to minimise the influence of outliers, focusing on median values, rather than means. Consistent with this practice, we used quantile regression (rather than OLS) when estimating transfer functions[3]; coefficients were used in conjunction with data from ABS (relating to NRM area, population density and incomes/regional product) to develop equations to contextualise (sub) ES value estimates for each NRM. Appendix D lists the studies and values used in this analysis; it also provides results from the regressions and equations used to contextualise values by NRM.

2.3.4 Cultural Services

Haines-Young and Potschin (2012) segregate cultural services into two divisions, which we discuss separately. In line with recommendations from Díaz *et al.* (2018), we further contextualised our estimates, including a category for Aboriginal and Torres Strait Islander (Indigenous) cultural values.

Cultural Services I: Use-values

We focused on aesthetic, amenity and recreational values. Aesthetic values are often estimated using hedonic pricing techniques – but property prices also reflect amenity and recreational opportunities, so estimates generated from these studies may not always exclusively relate to aesthetic values. Similarly, studies of recreation values often employ the travel cost method which can capture recreation, aesthetic and amenity values. We thus treat all aesthetic, amenity and recreational values as if they capture a single 'use value', rather than treating as separable and adding (recognizing that in carefully designed studies it is possible to do so).

As previously, we consulted van der Ploeg and De Groot (2010)'s database, identifying studies that had generated estimates of use-values. This database reports most values on a per-hectare basis, but recreation use values are almost always estimated using methods that rely on per-person (or per household) expressions of value/utility. In line with Wei *et al.* (2018), we thus chose to work with the per-person estimates. When only per-hectare estimates were recorded in the database, we sourced original studies to identify per-person estimates where available. We also conducted additional searches for more recent estimates. We converted all estimates to 2015 AUD (per person, per annum), and determined which biome they related to. We also compiled information on size of the study site, population of country, area of country, and GDP per capita (for the year in which the study had taken place). If there was insufficient information provided in the primary study, we sought such information from other sources (FAO, 2019; The World Bank, 2019).

---

[3] We also ran models using OLS regression; the sign and magnitude of coefficients were similar to those reported here (within about 10%-20%) but generally larger. P-values were generally lower (with statistically significant coefficients for most models)

Using this sample of estimates (N=136), we estimated transfer functions using quantile regressions; coefficients were used in conjunction with ABS data to generate contextualised estimates of per-person use-values associated with each biome within each NRM. Use values are relevant to both residents and tourists, however, there is a danger of significant double counting if one first counts a resident's use-value within the NRM in which they normally reside, and then also counts that value, when the person visits another NRM. This is because a person who is in enjoying the use-values on offer in region A, cannot simultaneously be enjoying the use-values of region B. In theory, it would be possible to avoid double counting if domestic tourism data recorded NRM of origin as that data could be used to estimate 'net' tourism in each NRM, though, the agencies responsible for collecting tourism data do not use NRM regions. Even if such information were available, it would be a non-trivial exercise to derive net-tourism estimates, from it. We thus took the pragmatic step of estimating use values for residents and international tourists separately – flagging domestic tourism numbers as something that could be refined in in the future. Specifically:

1. We assumed that all resident use values are associated with the NRM in which a person lives. Total residential use values in each NRM were simply estimated by multiplying per person values by population estimates.
2. We used published data to estimate international tourist numbers for each NRM, multiplying those estimates by per-person use values.

Total use-values within each NRM were estimated by adding resident and tourist estimates. Appendix E lists the studies and values used in this analysis; it also provides results from the regressions and equations used to contextualise values by NRM.

Cultural services II: Non-use values

There were relatively few non-use value estimates in van der Ploeg and De Groot (2010) so we sourced most estimates from elsewhere – focusing mainly, although not exclusively, on research undertaken in Australia. As previously, we converted all estimates to 2015 AUD, wherever possible recording (a) value-per hectare; (b) value per person; (c) study area; and (d) population density for the country in which the study area was located. Rather than recording the GDP per capita of the country in which the study site was located, we recorded the GDP per capita of the country in which study-respondents lived (irrespective of the location of the study site), since it is household income which will constrain willingness to pay. Where a single study generated multiple estimates for one grouped biome (e.g., different types of grasses), we included the average of those estimates. In many cases, researchers sought to estimate non-use values for areas that included multiple biomes (e.g., the non-use values of a national park that has within it, areas of forest, woodland, grassland and lakes) – these were coded as covering 'multiple biomes'. Non-use studies often use choice-experiments to estimate the value of particular 'attributes' (e.g., water quality, number of endemic species). If no information was provided regarding the separability of attributes, we averaged the reported attribute-values before recording; where separability was established, we added.

As previously, we used data from this compilation of 69 values to estimate the transfer functions, regressing logged estimates of per-hectare non-use values against (logged) estimates of 'abundance' (study-site-area/country area), per-capita GDP and the population density of the country in which the study had been undertaken (for the relevant year)[4]. The functions were used in conjunction

[4] We also explored options for using per-person values; see supplementary materials for an explanation.

with data from ABS (relating to NRM area, population density and regional product) to contextualise value estimates for each NRM. Appendix F lists the studies and values used in this analysis; it also provides results from the regressions and equations used to contextualise values by NRM.

Cultural Services III: Indigenous cultural values

The inextricable link between Indigenous cultural values and the health/condition of land and sea country (referred to simply as 'country') is widely documented (Hill *et al.*, 2011), but rarely quantified or valued in monetary terms. A decision to do so, is not uncontroversial. At least part of the issue is related to the appropriateness of methods, some is epistemological and some is ethical. For numerous reasons stated preference valuation techniques, which are based on the construction of hypothetical markets, are not generally useable in Indigenous settings (Venn & Quiggin, 2007; Farr *et al.*, 2016a; Stoeckl *et al.*, 2018). We have conferred with several Indigenous scholars when considering valuation in another study (for the Great Barrier Reef) – and these scholars gave 'in principle' support to the idea of including very rough estimates in a broader assessment, because to exclude them altogether may risk implicitly assigning them a value of zero. There is, however, collective agreement on the need to ensure that more detailed work is undertaken to properly assess these values, and to ensure that this work is led by Indigenous scholars.

We used information from the relatively small number of relevant Australian studies that we were able to find, to draw inferences about potential values, concluding that Indigenous cultural values would – at the barest minimum – amount to about $3100 per person per annum[5]. We consider this a 'place-holder' – used until a more appropriate method of allowing for these crucially important values is developed. Multiplying $3100 by the estimated number of Indigenous people living on mainland Australia gives a collective 'place holder' of $1.9781b per annum. Actual values are likely to be much higher than these – because many Indigenous cultural values are not even partially substitutable for other goods/services. In principle, we could spatially allocate these values down to the NRM level – multiplying estimates of the number of Indigenous residents by the per-person values. But many Indigenous people live away from their traditional lands. Like non-Indigenous cultural non-use values, Indigenous cultural values are thus best considered as place-based, and thus arguably better recorded on a per-hectare basis. We therefore divided our estimate of total value ($1.9871b) by the total area of our NRMs to infer per-hectare values ($2.59 per hectare). We then multiply per-hectare values by the size of each NRM to generate NRM-specific values. Appendix G provides a more complete discussion of the studies used to inform these estimates.

2.3.5 Companion animals

Following the lead of O'Sullivan (2012), we consider different groups of animals/types of industries: (1) animals kept primarily as household pets (including horses for 'recreational/social' purposes and animals such as working farm dogs); and (2) animals associated with the racing industry (greyhound dogs and horses). We were unable to find economic data on the racing industry (and associated breeding activities) that could be allocated spatially so focus entirely on domestic pets and horses kept for recreational purposes only. To the best of our knowledge, no Australian researcher has generated pure [human] welfare estimates associated with these animals, so we use measures of 'value' that are comparable to those used for portfolio industries – namely expenditure (noting that

---

[5] Median of estimates from: Sangha *et al.* (2019), Taylor and Stanley (2005), the midpoint from Zander and Straton (2010), and the (mean) of all SVA estimates. Section G, supplementary materials, provides more background.

willingness to pay will exceed expenditure estimates, so – consistent with previous biases – the decision to use expenditure understates true values).

For cats, dogs and other small domestics animals, we use data from Animal Medicines Australia (2016) which reports average expenditure on pets per (owner) household, and pet ownership by state to generate estimates of the expected expenditure per capita on different pets, for different states (expenditure per owner household * % of households owning * average household size). This allows us to allocate values across the landscape using ABS census data on usual place of residence.

We use information about (a) ownership; (b) expenditure; and (c) stabling, to generate spatially explicit estimates of expenditure on recreational horses. O'Sullivan (2012) suggests that about 2% of Australians own (recreational) horses. We assume a constant rate of horse ownership across the entire country. Macleay (2018) found that more than 70% of owners kept their horses within 5km of their place of residence so we allocate values spatially according to the distribution of population. We use Gordon (2001)'s estimates of the cost of keeping horses for recreational purpose only: $8774 in metropolitan areas (converted to 2015 AUD); and $1500 for horses kept permanently in pasture. We use different annual costs for regions that are differentially 'remote', effectively assuming that expenditure falls by 5.53% for each one-point increase in the ARIA+. Appendix H provides a more complete discussion of the data and calculations underpinning these estimates.

2.3.6 Physical Infrastructure

We focus on the values associated with infrastructure such as buildings, roads, wharfs, pipes and wiring; and goods such as wooden furniture that may be susceptible to damage. We use estimates of net capital values, from the ABS experimental estimates (ABS, 2017), focusing on just a subset of all types of capital. This subset includes capital that is most likely to be susceptible to damage by invasive species (Lovell *et al.*, 2006; Burnett *et al.*, 2008; Arthur *et al.*, 2015) - namely dwellings and infrastructure associated with electricity and water utilities. Even for that subset of capital, we note that not all will be susceptible, so we consider just a fraction of the total (24%)[6]. We annualise estimates, and divide through by state population, to derive an estimate of the average per-capita value of dwellings and utilities 'at risk', for each state. We then exploit some mathematical tautologies to convert the state-wide per-capita estimates into NRM-specific estimates of the value per hectare and total value of utilities 'at risk' (using ALUM data on utilities in each NRM). Appendix I provides a more complete discussion of the data and calculations underpinning these estimates.

2.4 Vulnerability

We then turned our attention to the question of vulnerability. That is, how vulnerable are these assets to the presence of the various biosecurity hazards? As we highlighted earlier (section 2.1), when seeking to value a single intervention to control a single pest/threat, it is simpler to directly estimate the damage (or willingness to pay (WTP) to avoid the damage) that might be caused by a hazard (using, for example, choice modelling) than it is to estimate both the value of the asset and the reduction in the value of the asset that would occur given the presence of the hazard (Table 1).

[6] The national accounts show that in 2017, about 24% of the capital in the electricity, gas and water sector was industrial machinery and equipment that could potentially be damaged by pests. We are unaware of any Australian study from which we could determine the percent of housing stock is 'at risk' (to pests), but note that in the US Guillebeau, Hinkle, and Roberts (2008) report that quotes for repairs from termite damage range from about $18,500 to $129,000 (AUD, 2015). This is between 6 and 40% of the median value of houses in their study area. In the absence of other information, we thus use the same estimate of infrastructure 'at risk' for housing, as we do for utilities.

As such, most studies that estimate the impact of invasive species do so using the former method. But on their own, these absolute (dollar) estimates of impact were insufficient to use as the basis for a whole-of-system valuation (Table 3) – and, discussed in section 2.1, even if there were estimates of the potential impact of all hazards on all assets, to naively add would be to risk double counting.

Instead, we use available studies of absolute damage (column III in Table 1) to estimate the relative damage to total asset values (column I in Table 1; expressed as a percent of the total) that different hazards are reported to have caused. These were then grouped by both asset type (e.g., regulating services) and hazard category (e.g., invertebrates) to determine the mean, median and maximum damages associated with each interaction. This is an attractive proposition for two reasons: firstly, it is possible to directly measure a relative change in some assets (such as percentage reduction in agricultural production; or reduction in water runoff); and secondly, it allows species' impacts to also be considered relative to others in the same hazard category (allowing their per unit area impacts to be inferred; potentially resolving the knowledge gaps preventing us from completing our valuation).

Damages to portfolio industries (i.e., agriculture and forestry) were excluded from our analysis as these impacts are largely directly observable and, therefore, relatively well described in the literature; see, for example, Hafi and Addai (2014) and Hafi *et al.* (2014) who estimate the market impacts of the 40 functional groups of taxa that we use also as the basis for our analysis, here.

2.4.1 Absolute vs Relative Vulnerability

Returning to the dataset that we compiled for our exploratory analysis in section 2.2, we looked for studies that report on empirical damages or WTP to avoid damages, categorising each according to both the type of hazard considered (often more than one) and the type of asset considered (often more than one). If damage estimates were reported separately for different hazard/asset groups, we recorded estimates separately, noting to which hazard/asset group each estimate referred. If damage estimates were reported at a more aggregated level, then we recorded estimates as a single value, listing the various assets or hazards considered (if more than two, listing 'multiple').

Some studies reported estimates of both current asset values and damages. If they did this, we simply used published estimates to calculate damages as a percent of current asset values. For studies that did not (or could not) estimate current values, we converted reported damages or WTP estimates to units that could be validly compared to the current value estimates generated in the first part of this study. To enable comparison, we first converted damage estimates to the same currency and year used when reporting current values. We then took one of two approaches:

1) *If damage estimates were reported as $ per-person*, we focussed on per-person current value estimates from part one of our study. Where possible we tried to match geography (e.g., if a damage study focused on NSW or QLD, we used only NRM data from NSW or QLD). We also tried to match 'biomes' (e.g., if a damage study reported estimates in wetlands, we only looked at current wetland values). If the damage study reported, say, damages to recreation use values but did not report biome specific estimates, then we calculated comparable per-person current value estimates by adding all recreation use values of Australian NRMs (irrespective of biome) and then dividing through by the Australian population to arrive at an 'average' estimate of current recreational values, per Australian. We then divided per-person damage estimates by our per-person current value estimates, to estimate damages as a percent of current values (i.e., relative impacts).

2) *If damage estimates were reported as per hectare*, we focused on per-hectare current value estimates from part one of our study. Where possible we tried to match geography (e.g., if a damage study focused on NSW or QLD, we used only NRM data from NSW or QLD). We also tried to match 'biomes' (e.g., if a damage study reported estimates in wetlands, we only looked at current wetland values). If the damage study reported, say, damages to recreation use values but did not report biome-specific estimates, then we calculated comparable per-hectare value estimates by adding all recreation use values of Australian NRMs (irrespective of biome) and then dividing through by the total numbers of hectares in Australia, to arrive at an 'average' estimate of current recreational values, per hectare. We then divided per-hectare damage estimates by our per hectare current value estimates, to estimate damages as a percent of current values (i.e., relative impacts).

2.4.2 Inferring Species Impacts

Chesson et al. (2014) and Parsons and Arrowsmith (2014) report the outcomes of several workshops whereby expert participants were asked to think about the potential 'consequence' of various hazards – beyond the more frequently assessed impacts on portfolio industries. For each of 55 different hazards participants were asked to rate the likely severity and extent of impact, on a scale of 0 (no potential consequence) to 4 (significant, nation-wide consequence) on ten different non-market assets. As discussed in the introduction, while the ABARES [RRRA] asset classification system works well for the purposes it was expressly developed for, using it as a classification system for generating aggregate estimates of 'value' risks double counting.

Consequently, as a first step, we re-classified impacts to ensure separability using Table 8 as a concordance guide. When more than one Likert measure was associated with an asset category (e.g., regulating services), we used the mean value. This left us with Likert scores for 40 functional groups of hazards (once the aquatic and zoonotic species were removed) for 5 different asset types (once social and human capital were removed). Although other studies (Hurley *et al.*, 2010; Liu *et al.*, 2011; Liu *et al.*, 2012; Measey *et al.*, 2016; Rumlerová *et al.*, 2016), have also used Likert-type scales to assess impacts, significant differences in approaches and in the asset categories precluded us from blending their insights with these. However, as outlined in the discussion, it is possible to use insights from other studies to judge the plausibility of the estimates.

The simplest approach to converting these Likert scores into % damage estimates would be to assume that a linear relationship exists between two variables. That is, those species with the maximum Likert score (4) would be assigned the maximum % damage estimate from section 2.4.1, with the midpoint (2) being assigned 50% of the % estimate, and so on. However, there are several clear limitations with this approach. Principally, participants in the original workshops envisaged a logarithmic relationship (base unspecified) between impact and the Likert score; that is, an up to order of magnitude difference between the impacts associated with each category (e.g., 1=0.1%, 2=1.0%, 3=10%.). Vulnerability also varies by both hazard and asset (i.e., the maximum damage caused by a plant pathogen on tourism is not equal to the to the maximum damage caused by a vertebrate on carbon sequestration. Consequently, we used a logistic function to transform the Likert scores where the midpoint was set to 2 (the middle Likert score), the asymptote was set to the maximum % estimate, by asset type, by hazard class and the steepness (reflecting the base) was set to 2 (which resulted in the best alignment between the mean observed and predicted impacts). Where we lacked asset-specific estimates, we used the 'multiple asset' estimate. As a final check we compared the means and variances of the observed and transformed estimates for similarity.

A note of caution – these estimates of vulnerability describe the average damage that is likely to occur, across an area, once a species from within that group has established. Depending on the scale of the analysis, it may take time for damages to rise to those levels (the temporal growth to the maximum value, potentially also following a logistic pattern) (Cook *et al.*, 2011; Cook *et al.*, 2012) and different species (even from within a particular group) will likely have different impacts (Aukema *et al.*, 2011; Paini *et al.*, 2016). So, while the estimates generated using this method are likely robust enough to be used to generate broad estimates of expected values (e.g., Paini *et al.*, 2016) they cannot not be used to predict actual impacts for individual species or to predict actual impacts at a particular point in time. Wherever possible, the inferred damages (estimated from these conversions) should be compared, and where necessary calibrated, with estimates generated from other empirical studies. Testing the sensitivity of final value estimates, to assumptions made about damage functions, is also critically important; particularly when knowledge gaps preclude the option of comparing and calibrating damage functions/estimates.

3 Results

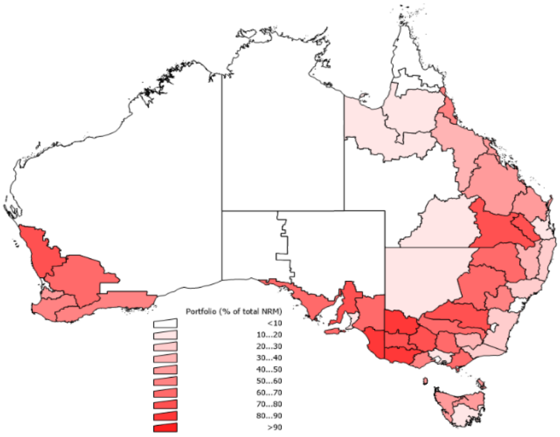
3.1 Values

Table 4 collates estimates of the flow of benefits in Australia. Even omitting two crucially important assets (human health and social capital), these add to approximately $250b per annum with a range of $174b to $1365b if using per unit values for regulating and cultural services associated with the lower and upper quartile of studies included in the assessment. Across all of Australia, less than half of those assets are closely associated with the market (including portfolio assets, infrastructure and expenditure on companion animals). Almost 60% of the assets protected are non-market 'values' – environmental goods and services which make significant contributions to social welfare (human wellbeing), but which are not generally bought and sold in the market and so do not always have attached to them, an explicit price. This is in line with other research (see, for example, Costanza *et al.*, 1997; Wei *et al.*, 2018), and highlights the importance of considering both market (agriculture and forestry) and non-market values for whole-of system assessments.

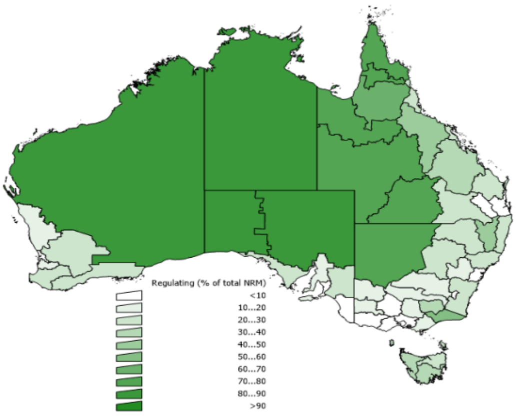
There is also much evidence of spatial heterogeneity (Figure 2). In accordance with intuition, the asset values that are most closely associated with the market are much more important (when considered as a percent of total values) to NRM regions in the southeast and southwest than elsewhere in Australia (market-related values are shown in the left panels of Figure 2). These are generally the NRM regions that have the highest population densities (Figure 2). The 'non-market' assets that are most closely associated with the environment (regulating services, water and subsistence values shown in the top two right maps of Figure 2) are relatively more important to the sparsely populated NRM regions than elsewhere. Cultural service values are generally most 'valuable' (relative to total services) in NRMs with relatively dense populations. NRM-specific estimates of the value of services provided by each asset are provided in Appendix J.

Table 4: The current value of assets protected by Australia's biosecurity system - $m, p.a. (equivalent to estimates of A1 and B1 from Table 1)

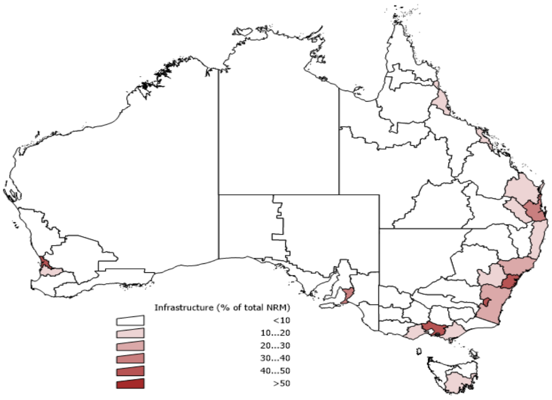
Relevant capital	Asset Broad class	Sub-class	Additional information	Estimated current value ($m p.a.)	Range ($m p.a.) – estimated from values associated with lower and upper quartiles of studies used in transfer equations, where available Q1	Q3
Natural	Provisioning-(portfolio)	Agriculture	Grazing	$27,398	$27,398	$27,398
			Cropping and horticulture	$3,2995	$3,2995	$3,2995
		Forestry (log production)	Plantation	$2,181	$2,181	$2,181
			Native forests	$11	$11	$11
		Aquaculture and fishing	*Omitted (data deficiencies and exceedingly difficult modelling challenges)*			
		TOTAL		$62,585	$62,585	$62,585
	Provisioning (non-portfolio)	Indigenous subsistence		$120	$120	$120
		Water	Drinking and purification	$16,232	$1,042	$78,609
		TOTAL		$16,353	$1,162	$78,783
	Regulating services	Mediation	Soil and water	$3,353	$1,164	$14,889
		Flood mitigation		$20,870	$2,562	$724,846
		Erosion prevention		$44,653	$40,249	$136,093
		Gene-pool/nursery		$19,841	$1,519	$90,786
		Carbon sequestration		$22,876	$15,772	$130,044
		TOTAL		$111,593	$61,264	$1,096,657
	Cultural services	Use values (recreation, aesthetics)	Australian residents	$8,298	$2,816	$29,933
			International tourists	$6,911	$2,917	$23,813
		Non-use values	Australian residents	$2,656	$498	$30,711
		Indigenous cultural values	PLACEHOLDER value only	$1,979	$1,979	$1,979
		TOTAL		$19,845	$8,209	$86,435.39
	Companion animals	Dog and Horse-racing & animals kept for breeding and racing	*Omitted – (data deficiencies and exceedingly difficult modelling challenges)*			
		Domesticated animals (excluding horses)		$11,723	$11,723	$11,723
		Horses for recreation and (non-racing) events		$3,525	$3,525	$3,525
		TOTAL		$15,247	$15,247	$15,247
Physical	Infrastructure	Dwellings	24% of net capital stock (annualised)	$21,746	$21,746	$21,746
		Utilities (electricity)		$4,152	$4,152	$4,152
		TOTAL		$25,898	$25,898	$25,898
Human	Human health	*Omitted – (data deficiencies and exceedingly difficult modelling challenges)*				
Social	Social-capital	*Omitted – (data deficiencies and exceedingly difficult modelling challenges)*				
Total				$251,519	$174,365	$1,365,605



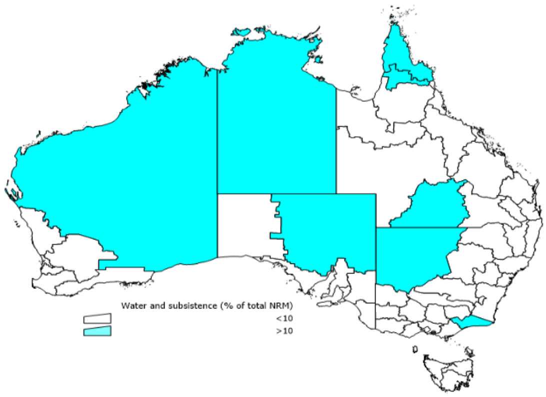
Income from portfolio industries



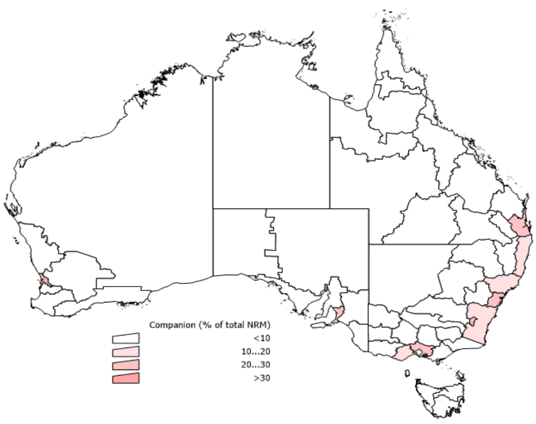
Regulatory service values



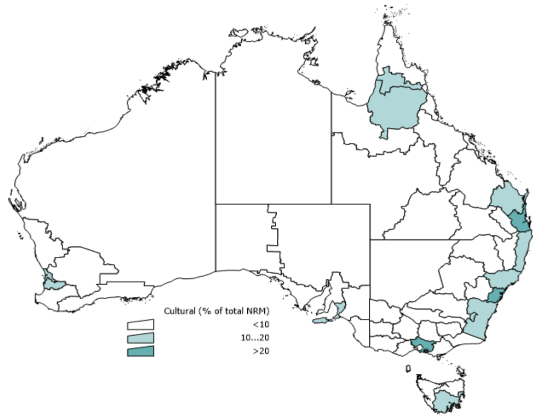
Infrastructure values (annualised)



Water and subsistence values



Expenditure on companion animals



Cultural values

Figure 2: Asset values as a percent of total assets (excl. human and social capital), by NRM (% of total asset value for NRM region).

3.2 Vulnerabilities

3.2.1 By Asset

Restricting our focus to studies for which it was possible to devise what we feel is a 'reasonable' comparison, we identified 41 studies (66 estimates) that included transferrable estimates of damages to non-portfolio assets (Table 5). We define a 'reasonable' comparison as being one where either the primary study provides enough data for us to directly estimate damages as a percent of current values, or one where damages relate to a specific hazard/asset group (facilitating a direct comparison) and where one could compare 'like with like' (e.g., values per person for infrastructure and most cultural values or values per hectare for particular types of biomes for regulating services). Studies that recorded damages greater than the value of the underlying assets were excluded from the dataset. The complete list of studies used in our analysis is included in Appendix K.

Table 5: Summary statistics relevant to studies from which it was possible to derive a reasonable estimate of the damages (% of total value) that biosecurity hazards are likely to impose on current assets.

Asset	N	Mean	Median	Min	Max
Subsistence	0				
Water & Regulating	9	9.89%	5.47%	0.04%	36.61%
Cultural	35	10.13%	7.58%	0.11%	38.44%
Infrastructure	5	3.33%	1.14%	0.03%	11.27%
Companion animals	0				
Multiple	17	1.70%	0.06%	0.00%	12.66%

Regulating (including water) and cultural assets had the highest average (mean and median) vulnerability to biosecurity hazards (c. 10%), followed by infrastructure (c. 3%; Table 5). As discussed earlier (section 2.2), we were unable to identify any studies with transferrable damage estimates for either of Indigenous subsistence or companion animals. Studies that estimated the impact of invasive species on multiple assets, recorded the lowest average vulnerability (c. 2%), likely reflecting the larger denominator in those circumstances. Maximum estimates of damage followed a similar trend, with cultural assets having the highest of the damage estimates (38%). The maximum estimates for each asset class were at least 3x the mean in all cases (Table 5).

3.2.2 By Functional Group

Table 6 provides a summary of the data from Table 9, showing the mean Likert (consequence) score by broad hazard group and asset category; portfolio assets were not included in these consequence assessments so are omitted here. Weeds had the highest mean score, followed by invertebrates, vertebrates, and pathogens. Though, significant variation existed within each of the taxonomic groups. In addition, these scores assume that each of the assets are equally weighted/valuable, which our earlier results (Table 4) confirm is not the case. Consequently, a high total score should only be interpreted as indicating that a species is likely to have a broad set of impacts (which may, or may not, be greater than a species that scores particularly highly against just a single asset).

Table 6: Mean Likert scores (scale 0-4) estimating the relative consequence (severity and extent) of an incursion, by broad taxonomic group and asset category.

	Provisioning non-portfolio	Regulatory services	Cultural services	Companion animals	Physical capital
Animal Pathogen	**0.24**	**0.10**	**0.11**	**1.33**	**0.00**
Invertebrate	**1.11**	1.07	1.56	0.52	0.54
Plant Pathogen	**0.85**	0.78	1.50	0.11	0.00
Vertebrate	**1.50**	1.13	0.00	1.00	0.00
Weed	**2.08**	1.63	1.50	0.75	0.00
MEAN	0.99	0.85	1.24	0.64	0.34

Of the assets, cultural and provisioning services had the highest relative vulnerability (Table 6) – similar to the results of our literature search (Table 5). In fact, the order of the assets' respective vulnerabilities was identical (e.g., A > B > C), and their relative vulnerabilities approximately similar (e.g., A = 1.2B = 3.0C). This suggests that the expert elicitation conducted by Parsons and Arrowsmith (2014) was likely successful in its aim to correctly identify the relative consequences of the various hazards that they sought to assess. Once transformed to percent/proportional reductions in asset yields (Table 7) we were then also able to confirm good absolute similarity with the empirical estimates gathered during the exploratory phase of the project (Table 5). That is, the mean vulnerability estimates for each asset were similar in both datasets – giving us some confidence that our estimates are plausible, notwithstanding our earlier caveats about the method's limitations.

Table 7: Mean reduction in the value of asset flows (% of total) in the event of an incursion, by broad taxonomic group and asset category.

	Provisioning non-portfolio	Regulatory services	Cultural services	Companion animals	Physical capital
Animal Pathogen	**0.34%**	**0.41%**	**0.51%**	**3.48%**	**0.00%**
Invertebrate	**2.94%**	7.49%	14.94%	1.06%	1.33%
Plant Pathogen	**2.08%**	5.23%	14.64%	0.17%	0.00%
Vertebrate	**3.40%**	5.42%	0.00%	1.51%	0.00%
Weed	**6.84%**	11.83%	11.90%	1.13%	0.00%
MEAN	2.58%	5.59%	11.38%	1.35%	0.85%

4 Discussion

When assessing values and vulnerabilities relevant to Australia's (or indeed any country's) biosecurity system it is essential to consider a broad range of impacts on a broad range of assets. Focusing on only a small subset of hazards or assets could unintentionally focus thought, policy and resources on that which is easy to estimate. However, this may not necessarily be that which is most important; it is crucially important to consider impacts 'at scale'.

But it is more than a little challenging to do this – primarily because most valuation methods have been developed from within the (sub) field of microeconomics and are thus designed to investigate changes and impacts at a small scale (i.e., they are *partial equilibrium* in nature). Moreover, despite the abundance of studies that consider economic issues relevant to biosecurity, the vast majority focus on market related impacts (Hafi *et al.*, 2015; Bradshaw *et al.*, 2016; Paini *et al.*, 2016) or consider just one or two pests (Burnett *et al.*, 2007; Gutrich *et al.*, 2007; Beville *et al.*, 2012) leaving significant gaps in the literature. From a compilation of 268 articles, we identified 117 separate studies that reported 262 estimates of damages associated with pests. But only 52 of those were Australian, more than 50% focused only on damages to [agriculture] portfolio industries, just 17% considered other non-market values such as those relating to regulating services, and there were no published estimates relating to the potential monetary value of incursions on 'Indigenous', 'social' or 'companion animals' asset categories.

These findings mirror those of Bradshaw *et al.* (2016) who also undertook an extensive search of literature relevant to the impacts of invertebrates. They identified no fewer than 737 studies – but only 21% (n=158) had useable economic estimates. They commented, in particular, on the absence of studies generating what they term 'reproducible' estimates of the impact of invertebrates on physical infrastructure and on regulatory services – the vast majority of studies being focused on costs/damages relevant to portfolio industries. Notably, Bradshaw *et al.* (2016)'s compilation did not identify any 'reproducible' empirical estimates of damages specific to the (non-portfolio) assets focused on here; we found only three. This points to a critical gap in our knowledge regarding the impacts of invasive species. That is not to say that the biophysical processes are not well understood – significant progress has been made in that regard in recent years – just that these studies aren't reporting impacts in a manner that enables their translation into economic analyses.

It is unsurprising, then, that data deficiencies prevented us from generating system-level estimates by compiling previously published values (and it is not feasible to estimate them all from scratch). Consequently, we chose to approach the whole-of-system valuation problem by determining first, the value of assets at risk (to biosecurity hazards); and second, the relative vulnerability of these assets to incursions – where data existed. We use observable market prices and a benefit transfer approach to do so. Benefit transfer is commonly used in the biosecurity literature, with researchers regularly compiling estimates (of losses in productivity, or control costs) generated in other studies, for use in theirs. See for example: Burnett *et al.* (2006); Burnett *et al.* (2007); and Burnett *et al.* (2008) in their studies of the impacts of miconia and brown tree-snakes on islands; Wylie and Janssen-May (2017)'s study of the impact of red imported fire ants in Australia; Xu *et al.* (2006)'s study of invasive species in China; and Pimentel *et al.* (2005)'s study of invasive species in the US.

Our study differs from these analyses in three key aspects: first, in that we use transfer functions rather than simple unit value transfers; second, in that we consider a broad range of assets that are ‘at risk’ from biosecurity hazards rather than just a single asset; and third, in that we examine the impacts of multiple hazards not just one. The most pressing challenge arising from these differences is the need to have a clear framework for aggregating together damages (sensu Parker *et al.*, 1999) to prevent both the multiple- and partial-counting of impacts (Holmes *et al.*, 2009; Boithias *et al.*, 2016). The CICES framework (Haines-Young & Potschin, 2012) is the obvious choice, though, its adoption to date in the biosecurity literature has been somewhat limited with more prominence given to the GISS/EICAT/SEICAT framework (e.g., Blackburn *et al.*, 2014; Nentwig *et al.*, 2016; Bacher *et al.*, 2018). There are a myriad of reasons for this; most notably (at least from our perspective) that the EICAT framework focusses on means outcomes where CICES is focussed on ends outcomes – and the disparate nature of the long-term monitoring of invasive species impacts (Pergl *et al.*, 2020) hinders the collection of data about ends. Nevertheless, we’re hopeful that our work in conjunction with the recently released INVACOST database (Diagne *et al.*, 2020) and possibly the Alien Scenarios project (Essl *et al.*, 2019) might stimulate more interest in this topic.

4.1 Asset Values

Notwithstanding our exclusions, we found that the biosecurity system protects assets which generate a flow of benefits worth in excess of A$250b p.a. (ranging from $174b to $1366b if using per unit values for regulating and cultural services associated with the lower and upper quartile of studies included in the assessment; Table 4). We are unaware of any other study that has sought to estimate the value of the same suite of assets across all of Australia, but note that at approximately A$325 p.a. per hectare, our estimate is significantly less than Costanza *et al.* (2014)’s estimated $4900 per hectare value associated with global terrestrial regions – or their earlier 1997 estimate of $1109 per hectare (Costanza *et al.*, 1997). Indeed, our estimate, which captures 16 different asset values, is less than the average per-hectare value estimates for some individual services (see, for example, the appendices where: Tables 11-14 report mean per hectare values associated with regulating services from 249 published estimates; Table 17 reports on cultural use values from 108 separate published estimates; and Table 21 reports on cultural non-use values from 33 separate studies). At least some of these differences are likely attributable to the fact that so much of Australia is arid and has low population densities. Most evident in the literature is that ecosystem service values (particularly regulating/maintenance services) are lower in arid regions than, for example, in forested areas. Use-values also depend on people: so uninhabited parts of the world will, by definition, have lower use values. But some of the difference between the broader global studies and our estimates are also methodological; whenever faced with a choice as researchers we have deliberately selected the option that generates an unambiguous downward bias. There is significant uncertainty in our estimates. But we can, at least, be confident that our estimates of current asset values, though admittedly imperfect and biased, are defensible. This is because we are certain of the direction of their bias: they unambiguously understate true values.

Most of the values protected by the biosecurity system are ones not normally captured through the market (i.e., they relate to regulating and non-use cultural services). This is despite the fact that more than 50% of the studies included in our biosecurity economics ‘stocktake’ considered only portfolio assets (i.e., those associated with agriculture, forestry and fishing) – these assets accounted for less than 25% of all values *at risk* across Australia (36% if using lower quartile values for non-market estimates, just 5% if upper quartile values). This suggests that the existing body of research may substantially understate the importance of biosecurity measures, particularly those that protect so called '*non-market'* assets, as has recently been suggested (Bradshaw *et al.*, 2016; Cuthbert *et al.*, 2020). In many respects this is not surprising, given that portfolio industries are arguably a provisioning service of nature, even though we do not typically think of them in that way. It also reflects findings from decades of studies worldwide, which repeatedly report that regulating and cultural services (most of which are not closely associated with the market) are often of greater value than provisioning services (see examples in van der Ploeg & De Groot, 2010).

Our NRM scale estimates of asset values begin to allow us to draw inferences about the potential 'vulnerability' of different regions to different types of hazards. Notably, areas inland from the major population centres appear most *at risk* to incursions that affect portfolio assets. The more populated regions near Australia's major cities have most assets at risk to hazards that impact infrastructure, companion animals, aesthetic and/or recreational values. Inland areas have a large proportion of *assets at risk* which are associated with regulating, non-use cultural, water and subsistence values and are thus likely to be most susceptible to hazards that cause environmental damage. Regional variations are even more evident when considering sub-classes of assets. Carbon sequestration values, for example, are driven solely by vegetation type with the highest values in mangrove, wetland, forest and woodland areas – regional variations in those values thus driven by regional differences in vegetation. Similarly contrasting patterns can be found in each of the 16 asset classes and 56 NRM regions, suggesting that the realised impact of pests/diseases is likely highly dependent on when and where an organism establishes in the first instance.

This suggests a broader benefit from adopting an asset led framework – namely that biosecurity hazards are not the only threat to these assets and, as such, our estimates have the potential to be used in a wide variety of contexts. One can easily imagine our estimates of water (provisioning) or carbon sequestration (regulating) being useful inputs into bushfire impact analysis and planning. Similarly, our estimates of flood mitigation (regulating) or coastal recreation (cultural) could be used in studies of climate change effects. Within the international disaster risk reduction literature (UNDRR, 2015) assets are referred to as 'exposures', though, our use of the terms 'hazard' and 'vulnerability' are otherwise consistent. This presents an opportunity for economies of scale. Non-market valuation is frequently cited as being expensive and difficult to do well (Bowen *et al.*, 2012; Hanley & Roberts, 2019). As we highlighted in the discussion, this leads to small-scale and piecemeal analyses. Having a more standardised approach would, therefore, enable more to be done for less. In the [much] longer term it also suggests the potential for a truly 'all agencies, all hazards' approach to disaster risk reduction and community resilience.

4.2 Asset Vulnerabilities

Returning to the reproducible estimates of asset vulnerability we are reminded of the sparseness of empirical data that can be reasonably be used to infer economic impacts. We were only able to identify 41 studies with (66) transferrable damage estimates of damages to non-market assets, over half of which related to cultural services (i.e., amenity, recreation and tourism; Table 5). Despite the relatively small sample size – critically – the ordinal rankings were identical in the two independently collected datasets (i.e., the empirical data and the expert elicitation data) giving us some confidence that our use of the expert elicitation data to fill gaps in the empirical estimates is sufficiently robust. The fact that our transformed estimates have a similar distribution to the empirical estimates is reassuring. In any case, our use of the 'multi-asset' vulnerability estimate to scale the elicited scores is a conservative choice, consistent with our approach to uncertainty throughout the entire study.

In comparison to the larger body of literature relating to portfolio damages these results also appear plausible. When adjusted for the size of their respective agricultural sectors, the various well-known estimates of total damage (summarised in Olson, 2006; Heikkilä, 2011; Marbuah *et al.*, 2014) suggest impacts in the range: 0.08 – 0.38 of agricultural yield. At the individual species level, the results of Hafi and Addai (2014); and Hafi *et al.* (2014) likewise indicate a similar range (min=0, mean=0.21, median=0.12, max=0.8), albeit with a greater maximum. Hayes *et al.* (2005) used non-monetary scales (0-1) to assess the potential economic, environmental and human health impacts of 53 marine pests. Averaged across all pests, their scores were also quite similar (minimums ≈ 0.1, mid-points ≈ 0.18, maximums ≈ 0.26) with the same ordinal relationships. Taken together, these comparisons suggest that our mean vulnerabilities in the range of 0.02-0.15 (Table 7) are likely well calibrated.

Of course, understanding the values of assets (or exposures) and their relative vulnerability to various hazards is just the first step in determining the expected impact of these hazards and the value of any risk reduction measures put in place to prevent them (Soliman *et al.*, 2015; Epanchin-Niell, 2017). The determination of impact also requiring knowledge about the likelihood of an outbreak of the various hazards occurring and the rate at which those hazards will spread across the landscape, encountering the various assets. Not to mention encountering each other. Given our stated aim of estimating the aggregate value of Australia's biosecurity system (Dodd *et al.*, 2017), our attention will now turn to these questions with a view towards estimating both the damages that one would expect to occur should the biosecurity system be turned off and those that will occur despite its operation – following the framework that we set out in Table 1 (Dodd *et al.*, 2020).

4.3 Limitations

Irrespective of our belief that our estimates are well calibrated it is critical that we acknowledge the many necessary limitations and assumptions upon which they are based. Mostly, these limitations arise due to significant knowledge gaps and data deficiencies forcing us to make assumptions or rely on expert judgement in lieu of empirical data. For example, as we have discussed extensively, there is a paucity of Australian studies that examine the impact of pests / diseases on assets other than agriculture, therefore, benefit transfer techniques must be relied on to obtain such data. Our approach to this has been clear – where sufficient data existed, we used that data to inform our inputs, but where it didn't, we omitted that element from our analysis. As such, our analysis does not consider impacts on social or human capital. Nor does it consider aquatic or zoonotic organisms. Whenever we transferred values, we used medians rather than means, therefore, minimising the influence of outliers. Similarly, wherever ambiguity existed about the assignment of a value to a group we always defaulted to the lower estimate. Whilst, in aggregate, these decisions will lower our overall estimate of value we believe that such an approach provides the most defendable result.

4.4 Conclusions

Biosecurity is defined as the minimisation of the adverse impact of pests and diseases on the economy, the environment and the community (COAG, 2019). In Australia, at least, biosecurity is most commonly contextualised as protecting Australia's A$62b agriculture sector (ABARES, 2018; ABS, 2018) from hazards such as foot and mouth disease that is predicted to have a $52b impact (Buetre *et al.*, 2013) if a widespread outbreak were to occur. Though, this paints a narrower picture of biosecurity than the definition suggests. Our results demonstrate that the total flow of benefits arising from assets vulnerable to biosecurity hazards is in excess of A$ 250 billion per annum. Further, several of these assets – particularly regulating and cultural services – are highly vulnerable to the introduction to invasive species. Through the development of a transparent and repeatable framework for compiling current-value estimates we have opened the door to a new way of estimating impacts at the macro scale – one that is not only crucially important when attempting to value biosecurity interventions at the system level but also other impacts providing opportunities for economies of scale and a truly integrated approach to disaster risk reduction.

5 References

ABARES (2017), *Australian forest and wood products statistics: September and December quarters 2017*, Canberra, <http://www.agriculture.gov.au/abares/publications/display?url=http://143.188.17.20/anrdl/DAFFService/display.php?fid=pb_afwpsd9abfe20180524.xml>.

ABARES (2018), *Agricultural Commodity Statistics 2018*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

ABS (2017), *5220.0 Australian National Accounts: State Accounts.* [2016-17], Australian Bureau of Statistics, https://www.abs.gov.au/AUSSTATS/abs@.nsf/allprimarymainfeatures/C428966227DE5843CA258346000E946F

ABS (2018), *7503.0 - Value of Agricultural Commodities Produced* [2016-17], Australian Bureau of Statistics, https://www.abs.gov.au/AUSSTATS/abs@.nsf/allprimarymainfeatures/38D85263FD1CC395CA2583EB0021F3C7

Akter, S & Grafton, RQ (2010), Confronting uncertainty and missing values in environmental value transfer as applied to species conservation, *Conservation biology*, **24**, 1407-17.

Animal Medicines Australia (2016), *Pet ownership in Australia*, Animal Medicines Australia, Barton, Australia.

Arthur, T, Summerson, R & Mazur, K (2015), *A comparison of the costs and effectiveness of prevention, eradication, containment and asset protection of invasive marine species incursions*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

Aukema, JE, Leung, B, Kovacs, K, Chivers, C, Britton, KO, Englin, J, Frankel, SJ, Haight, RG, Holmes, TP & Liebhold, AM (2011), Economic impacts of non-native forest insects in the continental United States, *PLoS One*, **6**, e24587.

Bacher, S, Blackburn, TM, Essl, F, Genovesi, P, Heikkilä, J, Jeschke, JM, Jones, G, Keller, R, Kenis, M, Kueffer, C, Martinou, AF, Nentwig, W, Pergl, J, Pyšek, P, Rabitsch, W, Richardson, DM, Roy, HE, Saul, W-C, Scalera, R, Vilà, M, Wilson, JRU & Kumschick, S (2018), Socio-economic impact classification of alien taxa (SEICAT), *Methods in Ecology and Evolution*, **9**, 159-68.

Baker, R & Ruting, B (2014), *Environmental Policy Analysis: A guide to non-market valuation*, Productivity Commission, Canberra.

Barbier, EB, Czajkowski, M & Hanley, N (2017), Is the income elasticity of the willingness to pay for pollution control constant?, *Environmental and resource economics*, **68**, 663-82.

Bateman, IJ, Carson, RT, Day, B, Hanemann, M, Hanley, N, Hett, T, Jones-Lee, M, Loomes, G, Mourato, S & Pearce, D (2002), Economic valuation with stated preference techniques: a manual, *Economic valuation with stated preference techniques: a manual.*,

Beale, R, Fairbrother, J, Inglis, A & Trebeck, D (2008), *One biosecurity: a working partnership.*, Department of Agriculture, Fisheries and Forestry, Canberra.

Beville, ST, Kerr, GN & Hughey, KF (2012), Valuing impacts of the invasive alga Didymosphenia geminata on recreational angling, *Ecological Economics*, **82**, 1-10.

Blackburn, TM, Essl, F, Evans, T, Hulme, PE, Jeschke, JM, Kühn, I, Kumschick, S, Marková, Z, Mrugała, A & Nentwig, W (2014), A unified classification of alien species based on the magnitude of their environmental impacts, *PLoS biology*, **12**, e1001850.

Boardman, AE, Greenberg, DH, Vining, AR & Weimer, DL (2011), *Cost-benefit analysis: concepts and practice*, Prentice Hall, Upper Saddle River, New Jersey.

Boithias, L, Terrado, M, Corominas, L, Ziv, G, Kumar, V, Marqués, M, Schuhmacher, M & Acuña, V (2016), Analysis of the uncertainty in the monetary valuation of ecosystem services—A case study at the river basin scale, *Science of the Total Environment*, **543**, 683-90.

Born, W, Rauschmayer, F & Bräuer, I (2005), Economic evaluation of biological invasions - A survey, *Ecological Economics*, **55**, 321-36.

Bowen, B, Chesson, J, Mazur, K & Buetre, B (2012), *Options for incorporating non-market impacts in the Biosecurity Risk Return Project*, Department of Agriculture, Fisheries and Forestry, ABARES.

Bradshaw, CJ, Leroy, B, Bellard, C, Roiz, D, Albert, C, Fournier, A, Barbet-Massin, M, Salles, J-M, Simard, F & Courchamp, F (2016), Massive yet grossly underestimated global costs of invasive insects, *Nature communications*, **7**, 12986.

Buetre, B, Kim, Y, Tran, Q & Gunasekera, D (2006), Avian influenza: Potential economic impact of a pandemic on Australia, *Australian Commodities: Forecasts and Issues*, **13**, 351.

Buetre, B, Wicks, S, Kruger, H, Millist, N, Yainshet, A, Garner, G, Duncan, A, Abdalla, A, Trestrail, C & Hatt, M (2013), *Potential socio-economic impacts of an outbreak of foot-and-mouth disease in Australia*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

Burnett, K, Kaiser, B, Pitafi, BA & Roumasset, J (2006), Prevention, eradication, and containment of invasive species: illustrations from Hawaii, *Agricultural and Resource Economics Review*, **35**, 63-77.

Burnett, K, Kaiser, B & Roumasset, J (2007), Economic lessons from control efforts for an invasive species: Miconia calvescens in Hawaii, *Journal of Forest Economics*, **13**, 151-67.

Burnett, KM, D'evelyn, S, Kaiser, BA, Nantamanasikarn, P & Roumasset, JA (2008), Beyond the lamppost: optimal prevention and control of the brown tree snake in Hawaii, *Ecological Economics*, **67**, 66-74.

Carbone, JC & Smith, VK (2013), Valuing nature in a general equilibrium, *Journal of Environmental Economics and Management*, **66**, 72-89.

Chesson, J, Mamun, E, Raphael, B, Walters, S & Whitworth, B (2014), *Assessment of 'non-market' impacts for the risk-return resource allocation project*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

COAG (2019), *Intergovernmental Agreement on Biosecurity*, Council of Australian Governments, Canberra.

Cook, DC, Fraser, RW, Waage, JK & Thomas, MB (2011), Prioritising biosecurity investment between agricultural and environmental systems, *Journal für Verbraucherschutz und Lebensmittelsicherheit*, **6**, 3-13.

Cook, DC, Liu, S, Edwards, J, Villalta, ON, Aurambout, J-P, Kriticos, DJ, Drenth, A & De Barro, PJ (2012), Predicting the benefits of banana bunchy top virus exclusion from commercial plantations in Australia, *PloS one*, **7**, e42391.

Costanza, R, d'Arge, R, De Groot, R, Farber, S, Grasso, M, Hannon, B, Limburg, K, Naeem, S, O'neill, RV & Paruelo, J (1997), The value of the world's ecosystem services and natural capital, *nature*, **387**, 253-60.

Costanza, R, de Groot, R, Sutton, P, van der Ploeg, S, Anderson, SJ, Kubiszewski, I, Farber, S & Turner, RK (2014), Changes in the global value of ecosystem services, *Global environmental change*, **26**, 152-8.

Craik, W, Palmer, D & Sheldrake, R (2017), *Priorities for Australia's biosecurity system: An independent review of the capacity of the national biosecurity system and its underpinning Intergovernmental Agreement.*, Department of Agriculture and Water Resources, Canberra.

Creedy, J & Wurzbacher, AD (2001), The economic value of a forested catchment with timber, water and carbon sequestration benefits, *Ecological Economics*, **38**, 71-83.

Cuthbert, RN, Bacher, S, Blackburn, TM, Briski, E, Diagne, C, Dick, JTA, Essl, F, Genovesi, P, Haubrock, PJ, Latombe, G, Lenzner, B, Meinard, Y, Pauchard, A, Pyšek, P, Ricciardi, A, Richardson, DM, Russell, JC, Simberloff, D & Courchamp, F (2020), Invasion costs, impacts, and human agency: Response to Sagoff 2020, *Conservation Biology*, **Online**, 1-4.

De Lange, WJ & van Wilgen, BW (2010), An economic assessment of the contribution of biological control to the management of invasive alien plants and to the protection of ecosystem services in South Africa, *Biological Invasions*, **12**, 4113-24.

DeFries, R, Pagiola, S, Adamowicz, W, Akcakaya, HR, Arcenas, A, Babu, S, Balk, D, Confalonieri, U, Cramer, W & Falconí, F (2005), Analytical approaches for assessing ecosystem condition and human well-being, *Ecosytems and Human Well-being: Current state and trends, by Millenium Ecosystem Assessment. Washington: World Resources Institute*,

Diagne, C, Leroy, B, Gozlan, RE, Vaissière, AC, Assailly, C, Nuninger, L, Roiz, D, Jourdain, F, Jarić, I & Courchamp, F (2020), InvaCost, a public database of the economic costs of biological invasions worldwide, *Scientific Data*, **7**, 277.

Diagne, C, Leroy, B, Vaissière, A-C, Gozlan, RE, Roiz, D, Jarić, I, Salles, J-M, Bradshaw, CJA & Courchamp, F (2021), High and rising economic costs of biological invasions worldwide, *Nature*, **592**, 571-6.

Díaz, S, Pascual, U, Stenseke, M, Martín-López, B, Watson, RT, Molnár, Z, Hill, R, Chan, KM, Baste, IA & Brauman, KA (2018), Assessing nature's contributions to people, *Science*, **359**, 270-2.

Dodd, A, Spring, D, Schneider, K, Hafi, A, Fraser, H & Kompas, T (2017), *Year 1 Report: Valuing Australia's Biosecurity System, CEBRA Project 1607A - Milestone 6*, Centre of Excellence for Biosecurity Risk Analysis (CEBRA), The University of Melbourne, Melbourne.

Dodd, AJ, Stoeckl, N, Baumgartner, JB & Kompas, TF (2020), *Key Result Summary: Valuing Australia's Biosecurity System*, Centre of Excellence for Biosecurity Risk Analysis (CEBRA), The University of Melbourne, Melbourne.

DoEE (2017), *Natural Resource Management (NRM) Regions* [2017], Australian Government Deprtment of the Environment and Energy, https://www.environment.gov.au/fed/catalog/search/resource/details.page?uuid=122fd267-e687-4465-9f2a-229953faf7d0

DoEE (2018), *NVIS Major Vegetation Groups (Version 5.1)* Australian Government Department of the Environment and Energy, http://www.environment.gov.au/land/native-vegetation/national-vegetation-information-system/data-products#key51

Epanchin-Niell, RS (2017), Economics of invasive species policy and management, *Biological Invasions*, **19**, 3333-54.

Essl, F, Lenzner, B, Courchamp, F, Dullinger, S, Jeschke, JM, Kühn, I, Leung, B, Moser, D, Roura-Pascual, N & Seebens, H (2019), Introducing AlienScenarios: a project to develop scenarios and models of biological invasions for the 21 st century, *NeoBiota*, **45**, 1-17.

FAO (2019), *FAOSTAT: Macro Indicators* Food and Agriculture Orgnisation of the United Nations, http://www.fao.org/faostat/en/#data/MK

Farr, M, Stoeckl, N, Esparon, M, Grainger, D & Larson, S (2016a), *Economic values and Indigenous protected areas across Northern Australia*, James Cook University, http://www.nespnorthern.edu.au/wp-content/uploads/2017/04/Economic_Values_and_IPAs_final_web.pdf.

Farr, M, Stoeckl, N, Esparon, M, Larson, S & Jarvis, D (2016b), The importance of water clarity to Great Barrier Reef tourists and their willingness to pay to improve it, *Tourism Economics*, **22**, 331-52.

Fitzpatrick, L, Parmeter, CF & Agar, J (2017), Threshold Effects in Meta-Analyses With Application to Benefit Transfer for Coral Reef Valuation, *Ecological Economics*, **133**, 74-85.

Getzner, M, Spash, C & Stagl, S (2004), *Alternatives for environmental valuation*, vol. 4, Routledge.

Gordon, J (2001), *The horse industry: Contributing to the Australian economy*, Rural Industries Research and Development Corporation, Canberra.

Gutrich, JJ, VanGelder, E & Loope, L (2007), Potential economic impact of introduction and spread of the red imported fire ant, Solenopsis invicta, in Hawaii, *Environmental Science & Policy*, **10**, 685-96.

Hafi, A & Addai, D (2014), *Economic consequences for species representing different pest groups affecting agricultural industries*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

Hafi, A, Addai, D, Zhang, K & Gray, EM (2015), *The value of Australia's biosecurity system at the farm gate: an analysis of avoided trade and on-farm impacts*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

Hafi, A, Morey, K & Zhang, Ky (2014), *Estimation of pest and disease consequence for portfolio industries: ABARES contribution to Risk Return Resource Allocation project*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

Hafi, A, Spring, D, Croft, L, Kompas, T & Morey, K (2013), Cost-effectiveness of biosecurity response options to red imported fire ants in South East Queensland, *Australian Bureau of Agricultural and Resource Economics report for the National Biosecurity Committee, Canberra, Australia*,

Haines-Young, R & Potschin, M (2012), Common international classification of ecosystem services (CICES, Version 4.1), *European Environment Agency*, **33**, 107.

Hanley, N & Roberts, M (2019), The economic benefits of invasive species management, *People and Nature*, **1**, 124-37.

Hayes, K, Sliwa, C, Migus, S, McEnnulty, F & Dunstan, P (2005), National priority pests: Part II Ranking of Australian marine pests, *An independent report undertaken for the Department of Environment and Heritage by CSIRO Marine Research*,

Heikkilä, J (2011), Economics of biosecurity across levels of decision-making: a review, *Agronomy for Sustainable Development*, **31**, 119-38.

Hill, R, Walsh, F, Davies, J & Sandford, M (2011), *Our country our way: guidelines for Australian indigenous protected area management plans*, CSIRO Ecosystem Sciences and Australian Government Department of Sustainability, Water, Environment, Population and Communities, Cairns.

Holmes, TP, Aukema, JE, Von Holle, B, Liebhold, A & Sills, E (2009), Economic impacts of invasive species in forests, *Annals of the New York Academy of Sciences*, **1162**, 18-38.

Hulme, PE (2009), Trade, transport and trouble: managing invasive species pathways in an era of globalization., *Journal of Applied Ecology*, **46**, 10-8.

Hurley, MV, Lowell, KE, Cook, DC, Liu, S, Siddique, A-B & Diggle, A (2010), Prioritizing biosecurity risks using a participatory decision-making tool, *Human and Ecological Risk Assessment*, **16**, 1379-94.

Johnston, RJ, Besedin, EY & Stapler, R (2017), Enhanced geospatial validity for meta-analysis and environmental benefit transfer: an application to water quality improvements, *Environmental and Resource Economics*, **68**, 343-75.

Kompas, T, Chu, L, Ha, PV & Spring, D (2019), Budgeting and portfolio allocation for biosecurity measures, *Australian Journal of Agricultural and Resource Economics*, **63**, 412-38.

Liu, S, Cook, D, Walker, I, Barry, S, White, B & De Barro, P (2014), *How do you measure biosecurity impact?*, Plant Biosecurity Cooperative Research Centre, Canberra.

Liu, S, Sheppard, A, Kriticos, D & Cook, D (2011), Incorporating uncertainty and social values in managing invasive alien species: a deliberative multi-criteria evaluation approach, *Biological Invasions*, **13**, 2323.

Liu, S, Walshe, T, Long, G & Cook, D (2012), Evaluation of Potential Responses to Invasive Non-Native Species with Structured Decision Making, *Conservation Biology*, **26**, 539-46.

Lovell, SJ, Stone, SF & Fernandez, L (2006), The economic impacts of aquatic invasive species: a review of the literature, *Agricultural and Resource Economics Review*, **35**, 195-208.

Macleay, CM (2018), 'The feeding and management practices of Australian horse owners', BVSc (Hons) thesis, Charles Sturt University.

Marbuah, G, Gren, I-M & McKie, B (2014), Economics of harmful invasive species: a review, *Diversity*, **6**, 500-23.

MEA, MEA (2005), Ecosystems and human well-being: synthesis, *Island, Washington, DC*,

Measey, G, Vimercati, G, Villiers, Fd, Mokhatla, M, Davies, S, Thorp, C, Rebelo, A & Kumschick, S (2016), A global assessment of alien amphibian impacts in a formal framework, *Diversity and Distributions*, **22**, 970-81.

Nairn, ME, Allen, PG, Inglis, AR & Tanner, C (1996), *Australian quarantine: a shared responsibility*, Department of Primary Industries and Energy, Canberra.

Nentwig, W, Bacher, S, Pyšek, P, Vilà, M & Kumschick, S (2016), The generic impact scoring system (GISS): a standardized tool to quantify the impacts of alien species, *Environmental Monitoring and Assessment*, **188**, 315.

O'Sullivan, H (2012), *What is the size and scope of the Australian horse industry and what does this mean for infectious disease management*, Australia: Australian Horse Industry Council (AHIC), Geelong.

Olson, LJ (2006), The economics of terrestrial invasive species: a review of the literature, *Agricultural and Resource Economics Review*, **35**, 178-94.

Paini, DR, Sheppard, AW, Cook, DC, De Barro, PJ, Worner, SP & Thomas, MB (2016), Global threat to agriculture from invasive species, *Proceedings of the National Academy of Sciences*, **113**, 7575-9.

Paracchini, ML, Zulian, G, Kopperoinen, L, Maes, J, Schägner, JP, Termansen, M, Zandersen, M, Perez-Soba, M, Scholefield, PA & Bidoglio, G (2014), Mapping cultural ecosystem services: A framework to assess the potential for outdoor recreation across the EU, *Ecological Indicators*, **45**, 371-85.

Parker, IM, Simberloff, D, Lonsdale, W, Goodell, K, Wonham, M, Kareiva, P, Williamson, M, Von Holle, B, Moyle, P & Byers, J (1999), Impact: toward a framework for understanding the ecological effects of invaders, *Biological invasions*, **1**, 3-19.

Parsons, S & Arrowsmith, L (2014), *Non-portfolio consequences of exotic organisms spreading in Australia and multiple incursion factor*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

Pascual, U, Muradian, R, Brander, L, Gómez-Baggethun, E, Martín-López, B, Verma, M, Armsworth, P, Christie, M, Cornelissen, H & Eppink, F (2010), The economics of valuing ecosystem services and biodiversity, *TEEB–Ecological and Economic Foundation*,

Pergl, J, Pyšek, P, Essl, F, Jeschke, JM, Courchamp, F, Geist, J, Hejda, M, Kowarik, I, Mill, A, Musseau, C, Pipek, P, Saul, W-C, von Schmalensee, M & Strayer, D (2020), Need for routine tracking of biological invasions, *Conservation Biology*, **Online**,

Pimentel, D, Zuniga, R & Morrison, D (2005), Update on the environmental and economic costs associated with alien-invasive species in the United States, *Ecological Economics*, **52**, 273-88.

Pyšek, P, Jarošík, V, Hulme, PE, Pergl, J, Hejda, M, Schaffner, U & Vilà, M (2012), A global assessment of invasive plant impacts on resident species, communities and ecosystems: the interaction of impact measures, invading species' traits and environment, *Global Change Biology*, **18**, 1725-37.

Pyšek, P, Richardson, DM, Pergl, J, Jarošík, V, Sixtova, Z & Weber, E (2008), Geographical and taxonomic biases in invasion ecology, *Trends in ecology & evolution*, **23**, 237-44.

Ricciardi, A (2007), Are modern biological invasions an unprecedented form of global change?, *Conservation Biology*, **21**, 329-36.

Rumlerová, Z, Vilà, M, Pergl, J, Nentwig, W & Pyšek, P (2016), Scoring environmental and socioeconomic impacts of alien plants invasive in Europe, *Biological invasions*, **18**, 3697-711.

Sangha, KK, Stoeckl, N, Crossman, N & Costanza, R (2019), A state-wide economic assessment of coastal and marine ecosystem services to inform sustainable development policies in the Northern Territory, Australia, *Marine Policy*, **107**, 103595.

Schlueter, M, McAllister, R, Arlinghaus, R, Bunnefeld, N, Eisenack, K, Hoelker, F, Milner-Gulland, E, Müller, B, Nicholson, E & Quaas, M (2012), New horizons for managing the environment: A review of coupled social-ecological systems modeling, *Natural Resource Modeling*, **25**, 219-72.

Soliman, T, Mourits, MCM, Oude Lansink, AGJM & van der Werf, W (2010), Economic impact assessment in pest risk analysis, *Crop Protection*, **29**, 517-24.

Soliman, T, Mourits, MCM, Oude Lansink, AGJM & van der Werf, W (2015), Quantitative economic impact assessment of invasive plant pests: What does it require and when is it worth the effort?, *Crop Protection*, **69**, 9-17.

Spring, D, Keith, JM & Kompas, T (2014), *Benefit-Cost Analysis of the National Four Tropical Weeds Eradication Program*, Australian Centre for Biosecurity and Environmental Economics Canberra.

Stoeckl, N, Farr, M, Larson, S, Adams, VM, Kubiszewski, I, Esparon, M & Costanza, R (2014), A new approach to the problem of overlapping values: A case study in Australia' s Great Barrier Reef, *Ecosystem Services*, **10**, 61-78.

Stoeckl, N, Hicks, C, Farr, M, Grainger, D, Esparon, M, Thomas, J & Larson, S (2018), The Crowding Out of Complex Social Goods, *Ecological Economics*, **144**, 65-72.

Tardieu, L & Tuffery, L (2019), From supply to demand factors: What are the determinants of attractiveness for outdoor recreation?, *Ecological Economics*, **161**, 163-75.

Taylor, J & Stanley, O (2005), *The opportunity costs of the status quo in the Thamarrurr Region*, Centre for Aboriginal Economic Policy Research, Australian National University, Canberra.

The World Bank (2019), *Countries and Economies Data* The World Bank, https://data.worldbank.org/country/

UNDRR (2015), *Sendai Framework for Disaster Risk Reduction 2015 - 2030*, United Nations Office for Disaster Risk Reduction, Sendai, Japan.

van der Ploeg, S & De Groot, R (2010), *The TEEB Valuation Database–a searchable database of 1310 estimates of monetary values of ecosystem services* Foundation for Sustainable Development, Wageningen, The Netherlands

Venn, TJ & Quiggin, J (2007), Accommodating indigenous cultural heritage values in resource assessment: Cape York Peninsula and the Murray–Darling Basin, Australia, *Ecological Economics*, **61**, 334-44.

Vilà, M, Basnou, C, Pyšek, P, Josefsson, M, Genovesi, P, Gollasch, S, Nentwig, W, Olenin, S, Roques, A & Roy, D (2010), How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment, *Frontiers in Ecology and the Environment*, **8**, 135-44.

Wei, F, Costanza, R, Dai, Q, Stoeckl, N, Gu, X, Farber, S, Nie, Y, Kubiszewski, I, Hu, Y, Swaisgood, R, Yang, X, Bruford, M, Chen, Y, Voinov, A, Qi, D, Owen, M, Yan, L, Kenny, D, Zhang, Z, Hou, R, Jiang, S, Liu, H, Zhan, X, Zhang, L, Yang, B, Zhao, L, Zheng, X, Zhou, W, Wen, Y, Gao, H & Zhang, W (2018), The value of ecosystem services from giant panda reserves, *Current Biology*, **28**, 1-7.

Williamson, M & Fitter, A (1996), The varying success of invaders, *Ecology*, **77**, 1661-6.

WTO (1994), *General agreement on tariffs and trade*, Geneva, Switzerland.

Wylie, FR & Janssen-May, S (2017), Red Imported Fire Ant in Australia: What if we lose the war?, *Ecological Management & Restoration*, **18**, 32-44.

Xu, H, Ding, H, Li, M, Qiang, S, Guo, J, Han, Z, Huang, Z, Sun, H, He, S & Wu, H (2006), The distribution and economic losses of alien species invasion to China, *Biological Invasions*, **8**, 1495-500.

Zander, KK & Straton, A (2010), An economic assessment of the value of tropical river ecosystem services: Heterogeneous preferences among Aboriginal and non-Aboriginal Australians, *Ecological Economics*, **69**, 2417-26.

6 Supplementary Tables

Table 8: Asset categories for whole-of-system monetary assessments of Australia's biosecurity system.

Type of 'capital'	Impact broad category	Impact sub-category
Natural Capital	Provisioning services Ecosystems provide food, water, energy and material goods (e.g. for housing, and other things)	**Portfolio industries** describe provisioning services that are sold in the marketplace – agriculture, fishing and forestry, including trade. Non-portfolio provisioning services are defined here as provisioning services which are not sold in the market or part of a portfolio industry. They include things such as the value of food which Indigenous people harvest (cultural values associated with the harvest are considered elsewhere) and other materials extracted from ecosystems which are used but not sold. Also included here, is water for drinking and (non-portfolio) industry use
	Regulating & maintenance services (R&M) Includes impacts on three asset categories used in RRRA model: ***Native biodiversity, regulating functions and atmosphere***	Sub-category definitions match those of CICES. Mediation of waste, toxics & other nuisances: For example, forests help purify the air by removing pollutants such as SO_2 and NOx (this is in addition to their ability to sequester carbon). Similarly, wetlands and seagrass beds help filter sediments from waterways, and remove nutrients – which can, in excess, cause damage elsewhere (nutrient loads in the GBR being a notable example). Mediation of flows – e.g. reefs, mangroves and other wetlands provide storm surge protection to people and physical infrastructure by slowing liquid flows. Maintenance of physical, chemical, biological conditions (includes ecological, soil, water and atmospheric quality). Natural habitats provide the required environment for species to survive and to pass-on genes from one generation to another, which maintains the continuity of ecological processes and functions, and hence ES. Vegetation (and other core parts of an ecosystem) helps maintain soil nutrients - a service which is valuable to ecosystems by and of itself, but which also benefits people (e.g. enhancing agricultural productivity). Ecosystems also stabilise atmospheric conditions – most tangibly by sequestering carbon.
	Cultural services Includes impacts on amenity (from RRRA model)	Sub-category definitions match those of CICES. Physical and intellectual interactions with ecosystems & land-/seascapes. Some of these cultural services are closely related to the market (e.g. those relating to tourism). Others are not. Aesthetic, amenity and lifestyle values are not traded in markets, but their value is often built in to other market prices. A house with an ocean view will generally sell for more than an identical house without one, and people who must live and work in unpleasant environments often need to be offered higher wages than those living (and working) in more attractive locales. Spiritual, symbolic & other interactions with ecosystems & land-/seascapes are rarely, if ever traded in the market.
	Domesticated and companion animals	Definition matches that used in the RRRA model assessments. Includes impacts on species not captured in the other categories (particularly portfolio industries, regulating and maintenance services and cultural services) (e.g. dogs, cats, horses, aquarium fish and aviary birds).
Human Capital	Human health	Definition matches that used in the RRRA model assessments. Measures consider the impact of pests or diseases on the physical health of residents.
Physical Capital	(Physical) Infrastructure and produced goods	Definition matches that used in the RRRA assessments. Includes the impact of the pest or disease on infrastructure such as buildings, roads, wharfs, pipes and wiring; goods such as wooden furniture and stored food products not covered under portfolio industries.
Social Capital	Social (capital/infrastructure): impacts on human individuals and communities	Definition loosely matches that used in the RRRA model assessments – albeit with minor variations relating to provisioning services; note also the exclusion of impacts on physical places of residence, captured in physical infrastructure and the exclusion of social interactions that involve the environment that are considered within our cultural values category (including recreational and amenity values). Includes, impacts on, access to services, freedom of movement and relationships with others.

Table 9: Originally elicited Likert scores estimating the impact (between 0-4) of each functional group on assets following the framework outlined in Chesson et al. (2014) and Parsons and Arrowsmith (2014) as used in RRRA.

#	Functional group	Biodiversity (Direct)	Biodiversity (Indirect)	Amenity	Regulating	Water	Atmosph.	Domestic animals	Infrastr.	Social
1	AGM	4	2	3	2	0	1	0	0	2
2	Animal other bacteria	0	0	0	0	0	0	1	0	1
3	Animal other micro other	1	0	0	0	0	0	2	0	1
4	Animal other virus	0	0	0	0	0	0	2	0	2
5	Avian virus	2	0	1	0	1	0	1	0	1
6	Broadacre bacteria	3	1	3	2	1	0	0	0	2
7	Broadacre beetle	2.5	2	2.5	2	1	0	1	1	1
8	Broadacre bug thrips mite	3	3	3	3	1	0	1	0	1
9	Broadacre fungus	3	1.5	3	2	1	1	1	0	2
10	Broadacre mollusc	3	2	2	2	2	0	1	0	1
11	Broadacre virus	1	0	1	0	0	1	0	0	0
12	Broadacre weed	3	3	2	0	1	0	1	0	0
13	FMD	0	0	0	0	2	0	2	0	3
14	Forestry beetle	1	0	2	1	0	0	0	2	0
15	Forestry fungus	3	2	2	2	1	0	0	0	3
16	Forestry nematode	0	0	2	0	0	0	0	0	2
17	Forestry termite	2.5	1.5	2	0	0	0.5	0	4	2.5
18	Forestry weed	3	3	2	0	0	0	0	0	0
19	Fruit fly	0	0.5	0	0	0	0	0	0	2
20	GAS	3	2	2	0	0	0	0.5	0	2
21	Horticulture bacteria	1	0	0	0	0	0	0	0	2
22	Horticulture beetle	1	1	1	1	0.5	0	0	0	2
23	Horticulture bug thrips mite	2	1	0.5	0	0	0	0	0	2
24	Horticulture fly moth	1	0.5	0.5	0	0	0	0	0	1
25	Horticulture fungus	1	0.5	0.5	0	0	0	0	0	2
26	Horticulture nematode	0	0.5	0	0	0	0	0	0	0.5
27	Horticulture virus	0.5	0.5	0	0	0	0	0	0	0.5
28	Horticulture weed	3	3	1	1	0	0	1	0	0
29	Khapra beetle	3	3	3	2	1	0	1	3	2
30	Livestock bacteria	0	0	0	0	0	0	0	0	1
31	Livestock bug thrips mite	0	1	0	3	1	0	1	0	0.5
32	Livestock fly moth	1	0	0	0	0	0	1	0	0
33	Livestock virus	0.5	0	0	0	0	0	1	0	1
34	Non-agricultural bee wasp	1	1	1	3	0	0	0	0	0
35	Non-agricultural fly moth	0	2.5	2	0	0	1	1	0	2
36	Non-agricultural fungus	0	1	2	0	0	0	0	0	2
37	Non-agricultural micro other	0	1	2	0	0	0	0	0	2
38	Non-agricultural vertebrate	3.5	1	0	0	0	0	1	0	1
39	Non-agricultural weed	3	3	1	1	0	0	1	0	0
40	Tramp ant	3	1	2	1	0	0	3	1	1

Table 10: Re-factored Likert scores (ensuring separability) estimating the impact (between 0-4) of each functional group on assets following the extended CICES framework.

#	Functional group	Provisioning (Non-Portf)	Regulating	Cultural	Domestic Animals	Infrastructure
1	AGM	2.00	2.25	3.00	0.00	0.00
2	Animal other bacteria	0.00	0.00	0.00	1.00	0.00
3	Animal other micro other	0.33	0.25	0.00	2.00	0.00
4	Animal other virus	0.00	0.00	0.00	2.00	0.00
5	Avian virus	1.00	0.50	1.00	1.00	0.00
6	Broadacre bacteria	1.67	1.50	3.00	0.00	0.00
7	Broadacre beetle	1.83	1.63	2.50	1.00	1.00
8	Broadacre bug thrips mite	2.33	2.25	3.00	1.00	0.00
9	Broadacre fungus	1.83	1.88	3.00	1.00	0.00
10	Broadacre mollusc	2.33	1.75	2.00	1.00	0.00
11	Broadacre virus	0.33	0.50	1.00	0.00	0.00
12	Broadacre weed	2.33	1.50	2.00	1.00	0.00
13	FMD	0.67	0.00	0.00	2.00	0.00
14	Forestry beetle	0.33	0.50	2.00	0.00	2.00
15	Forestry fungus	2.00	1.75	2.00	0.00	0.00
16	Forestry nematode	0.00	0.00	2.00	0.00	0.00
17	Forestry termite	1.33	1.13	2.00	0.00	4.00
18	Forestry weed	2.00	1.50	2.00	0.00	0.00
19	Fruit fly	0.17	0.13	0.00	0.00	0.00
20	GAS	1.67	1.25	2.00	0.50	0.00
21	Horticulture bacteria	0.33	0.25	0.00	0.00	0.00
22	Horticulture beetle	0.83	0.75	1.00	0.00	0.00
23	Horticulture bug thrips mite	1.00	0.75	0.50	0.00	0.00
24	Horticulture fly moth	0.50	0.38	0.50	0.00	0.00
25	Horticulture fungus	0.50	0.38	0.50	0.00	0.00
26	Horticulture nematode	0.17	0.13	0.00	0.00	0.00
27	Horticulture virus	0.33	0.25	0.00	0.00	0.00
28	Horticulture weed	2.00	1.75	1.00	1.00	0.00
29	Khapra beetle	2.33	2.00	3.00	1.00	3.00
30	Livestock bacteria	0.00	0.00	0.00	0.00	0.00
31	Livestock bug thrips mite	0.67	1.00	0.00	1.00	0.00
32	Livestock fly moth	0.33	0.25	0.00	1.00	0.00
33	Livestock virus	0.17	0.13	0.00	1.00	0.00
34	Non-agricultural bee wasp	0.67	1.25	1.00	0.00	0.00
35	Non-agricultural fly moth	0.83	0.88	2.00	1.00	0.00
36	Non-agricultural fungus	0.33	0.25	2.00	0.00	0.00
37	Non-agricultural micro other	0.33	0.25	2.00	0.00	0.00
38	Non-agricultural vertebrate	1.50	1.13	0.00	1.00	0.00
39	Non-agricultural weed	2.00	1.75	1.00	1.00	0.00
40	Tramp ant	1.33	1.25	2.00	3.00	1.00

Table 11: Estimated impact (proportional decline in the value of asset flows) of each functional group on assets following the extended CICES classification.

#	Functional group	Provisioning (Non-Portf)	Regulating	Cultural	Domestic Animals	Infrastructure
1	AGM	0.0633	0.227882	0.338578	0	0
2	Animal other bacteria	0	0	0	0.015091	0
3	Animal other micro other	0.004361	0.010731	0	0.0633	0
4	Animal other virus	0	0	0	0.0633	0
5	Avian virus	0.015091	0.017363	0.045822	0.015091	0
6	Broadacre bacteria	0.042948	0.098459	0.338578	0	0
7	Broadacre beetle	0.052847	0.117453	0.281019	0.015091	0.013434
8	Broadacre bug thrips mite	0.083652	0.227882	0.338578	0.015091	0
9	Broadacre fungus	0.052847	0.160287	0.338578	0.015091	0
10	Broadacre mollusc	0.083652	0.138218	0.1922	0.015091	0
11	Broadacre virus	0.004361	0.017363	0.045822	0	0
12	Broadacre weed	0.083652	0.098459	0.1922	0.015091	0
13	FMD	0.008225	0	0	0.0633	0
14	Forestry beetle	0.004361	0.017363	0.1922	0	0.05635
15	Forestry fungus	0.0633	0.138218	0.1922	0	0
16	Forestry nematode	0	0	0.1922	0	0
17	Forestry termite	0.02641	0.0542	0.1922	0	0.110673
18	Forestry weed	0.0633	0.098459	0.1922	0	0
19	Fruit fly	0.003155	0.008412	0	0	0
20	GAS	0.042948	0.066786	0.1922	0.006004	0
21	Horticulture bacteria	0.004361	0.010731	0	0	0
22	Horticulture beetle	0.011191	0.027772	0.045822	0	0
23	Horticulture bug thrips mite	0.015091	0.027772	0.018231	0	0
24	Horticulture fly moth	0.006004	0.013665	0.018231	0	0
25	Horticulture fungus	0.006004	0.013665	0.018231	0	0
26	Horticulture nematode	0.003155	0.008412	0	0	0
27	Horticulture virus	0.004361	0.010731	0	0	0
28	Horticulture weed	0.0633	0.138218	0.045822	0.015091	0
29	Khapra beetle	0.083652	0.18305	0.338578	0.015091	0.099266
30	Livestock bacteria	0	0	0	0	0
31	Livestock bug thrips mite	0.008225	0.04364	0	0.015091	0
32	Livestock fly moth	0.004361	0.010731	0	0.015091	0
33	Livestock virus	0.003155	0.008412	0	0.015091	0
34	Non-agricultural bee wasp	0.008225	0.066786	0.045822	0	0
35	Non-agricultural fly moth	0.011191	0.034907	0.1922	0.015091	0
36	Non-agricultural fungus	0.004361	0.010731	0.1922	0	0
37	Non-agricultural micro other	0.004361	0.010731	0.1922	0	0
38	Non-agricultural vertebrate	0.034048	0.0542	0	0.015091	0
39	Non-agricultural weed	0.0633	0.138218	0.045822	0.015091	0
40	Tramp ant	0.02641	0.066786	0.1922	0.111509	0.013434