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Full Length Article

The monetary value of 16 services protected by the Australian National Biosecurity System: Spatially explicit estimates and vulnerability to incursions

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ABSTRACT

Biosecurity systems protect numerous assets, distributed differentially across space. Focusing on Australia's 56 natural resource management regions, we generate spatially explicit estimates of the current value of 16 different services generated by *assets* that are protected by the biosecurity system (hereafter *values*). Benefit transfer functions are used to generate some *values*; others are derived from observable market data. Across all regions and services, we estimate an aggregate *value* of approximately $250b p.a. Nearly 90% of those *values* are ecosystem service values, associated with Australia's *Natural Capital* and more than one-half are services not normally bought or sold in the marketplace (e.g., a subset of cultural and most regulating services). We use insights from the literature, in conjunction with our *values,* to estimate the potential losses that (a) weeds and (b) invertebrates, could inflict in different regions – hereafter, v*ulnerabilities (*potential $ losses per hectare p.a.). Urban areas are generally more *vulnerable* than remote areas, and many regions are more *vulnerable* to invertebrates than weeds, but weed *vulnerabilities* dominate in several of the large, remote, NRMs across the north, in the 'outback' and in the west. Our *values* can be used to assess the vulnerability of natural capital, and other capitals, to a wide range of other threats and are thus of potential use in numerous policy settings. Our generic approach to considering impacts at large geographic scale (using *values* and then assessing *vulnerabilities)* is one that is useful and transferrable to other settings across the world.

1. Introduction

Those charged with developing biosecurity policy need information about the potential economic consequences of numerous biosecurity threats that could individually or simultaneously impact multiple 'assets' (be they economic, environmental, human or other) over time, and across space. The majority of economic studies that consider biosecurity issues focus on market related impacts (Bradshaw et al., 2016), but many also consider health or environmental impacts of pests and disease; fortunately, there are numerous valuation tools (Freeman III et al., 2014; Bennett, 2011; Pascual et al., 2010) available for use in non-market assessments. Researchers often approach large-assessment tasks by compiling estimates from multiple sources (through, for example, benefit transfer) and aggregating, but great care must be taken when doing so because compilation and aggregation issues are non-trivial; naively adding values to generate an estimate of multiple benefits can result in double counting (Boithias et al., 2016; Holmes et al., 2009).

Rather than estimating the total cost of a set of hazards by summing individual impacts, we suggest that system level impacts can alternatively be derived by first estimating the flow of benefits arising from assets protected by the system and second, considering the (potential) decline in the value of those assets that would occur with and without the system, assuming different probabilities of incursions. We provide an empirical demonstration of that approach, describing a systematic framework for considering issues relevant to Australia's (or indeed any nation's) biosecurity system. We estimate 16 different market and non-market values for 56 natural-resource management regions – benefit transfer functions are used for some values; others are directly derived from observable market data. We use the spatially differentiated value

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estimates to draw inferences about the vulnerability of different regions to different pests and diseases, noting that these value estimates could be used to assess the impact of a variety of different threats, above and beyond those relating to biosecurity.

We note that benefit transfer is commonly used in the biosecurity literature, with researchers regularly compiling estimates (of likely losses in productivity, or costs of control) generated in other studies, for use in theirs. See for example: K. Burnett et al. (2006); K. Burnett, Kaiser, and Roumasset (2007); K.M. Burnett et al. (2008) in their studies of the impacts of weeds and brown tree-snakes. See also: Wylie and Janssen-May (2017)'s study of the potential impact of red imported fire ants in Australia; Xu et al. (2006)'s study of invasive species in China; and Pimentel, Zuniga, and Morrison (2005)'s study of invasive species in the US. These studies use simple value-transfers and focus on just one pest (or type of pest) at a time. Our study differs from these, in that we use transfer functions and consider a broad range of assets that are 'at risk' from a variety of different types of pests. This is possibly the most comprehensive value assessment undertaken in Australia. Although the empirics are relevant only to Australia and focus on biosecurity as an application, our general approach – of considering first, the 'values at risk' (before considering impacts) – is one that is transferrable globally.

Our paper is structured as follows. Section two gives background information. It starts by providing relevant biosecurity context to appropriately frame the valuation task. It also provides a relatively extensive overview of relevant literature – presented not with the intent of describing the literature itself, but to highlight the significant knowledge gaps which led us to devise our novel approach of first estimating current values, and second considering impact. In section three we define the scope of our investigation before describing how we estimate current values and vulnerabilities, and then describing our scenario analysis. Results are presented in section four and discussed in section five. Our final section offers some concluding remarks.

2. Background

In Australia, Environmental Biosecurity is defined as 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. Formally, the term environment includes terrestrial, inland water and marine ecosystems, their constituent parts, and natural or physical resources, while the term Social Amenity includes social, economic and cultural aspects of the environment, including tourism, human infrastructure, cultural assets and national image (endorsed by NBC in May 2017). When assessing values and impacts relevant to Australia's biosecurity system it is thus essential to consider a broad range of impacts on a broad range of what we hereafter refer to as assets.

We ultimately aim to estimate the entire value of Australia's biosecurity system – but this is a non-trivial task, and so in this paper, we lay the foundations to later do so, while also generating both methodological and empirical insights. Evident from the literature is the fact that it is difficult to estimate the consequences of even a single biosecurity threat since impact mechanisms are numerous and varied (Blackburn et al., 2014). It is even more difficult to estimate the potential consequences of multiple threats on multiple assets. An additional layer of complexity is present here because Australia's biosecurity system is itself complex, with many inputs, actors/participants, activities and outputs (Dodd et al., 2017) and there are numerous ways in which the actions of various actors affect biophysical links within the system, the likelihood that any individual pest will enter, establish and spread, and the consequences of such incursions.

The tools used to assess costs and benefits associated with biosecurity threats and measures are numerous, varied, and complex. They include, but are not limited to bio-economic models, computer-based simulation models and/or statistical models which are calibrated with insights from local stakeholders and empirical research to generate a diverse range of value estimates, only a subset of which use money as the metric. See, for example: Lee et al. (2015) who use agent based models when considering the spread of influenza; Shepard et al. (2016) who examine the global prevalence of dengue; D.C. Cook et al. (2015) who assess the potential economic costs (to the cattle industry) of the mimosa invasion in the Kimberley and estimates of how much should 'optimally' be spent on eradication/ control programs; D. Cook et al. (2013) who estimate the losses likely to be incurred in the Northern Australian banana industry from yellow Sigatoka under different management guidelines; and Kompas et al. (2016) who consider the design of an optimal surveillance policy for weeds, using Australian Hawkweed as an example.

Recognising that most applied research has been undertaken at small scale, in that studies focus on a single threat/species and only estimate costs and benefits for a subset of people / actors / industries (e.g. on agricultural producers, or the agricultural industry), we sought to identify knowledge gaps by collating published papers and categorising monetary cost/benefit estimates according to the broad type of threat (pest) evaluated, the type of asset considered, and the type of value generated. Our aim was to determine which assets/threats we have most/least knowledge of.

For assets, we used categories that align with the Common International Classification of Ecosystem Services (CICES, Haines-Young and Potschin (2012) – albeit with minor variations to adequately cover all assets of concern to Australia's Biosecurity system, and to ensure categories are adapted for context (after Díaz et al. (2018)). We added a subcategory to provisioning services, hereafter referred to as 'portfolio industries' (agriculture and forestry) – the term used by the Department of Agriculture, Water and Environment when referring to key industries protected by Australia's biosecurity system. The term 'non-portfolio' is used to identify other provisioning services. We also included four additional assets/values identified by Biosecurity Australia (companion animals, physical infrastructure, human and social capital) – although were unable to find sufficient data to generate spatially explicit estimates for values associated human and social capital. Our final assessment thus focuses mainly, although not exclusively, on the 'values' associated with natural capital (Table 1).

For threats, we consulted the broader biosecurity literature. Seebens

Table 1
Capitals, Asset Types and Asset Classes used in the analysis.

Relevant Capital		Asset Type	Asset Class	Sub-class
Natural	Ecosystem Services	Provisioning	Portfolio Industries	Agriculture
				Forestry
			Indigenous Subsistence	Subsistence
			Water for Consumption	Water
		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
Built		Infrastructure	Dwellings / Utilities	Infrastructure

et al. (2017) recently found that no fewer than 16,926 species have established 'alien' populations outside their native range, globally. It is not feasible to account for all, but we note that several recent studies (e.g., Aukema et al., 2011; Epanchin-Niell et al., 2014) have classified these species into 'functional groups' according to their mode of action. This is justifiable since the impacts of species within a group and their management controls are generally highly similar. We thus sought to collate studies that reported estimates of the impact of pests or diseases (threats) on different types of assets, drawing on insights from other researchers (e.g. Akter and Grafton (2010) and De Lange and van Wilgen (2010)), to identify 'workable' categories of threats.

We identified 268 articles relevant to the economics of biosecurity. More than half did not report monetary estimates, so we focused on the 117 studies that did. We counted the number that provided empirical estimates of the potential (or actual) damages inflicted by each broad threat type on each asset category, and on control or eradication costs (Table 2). Some studies provided cost/benefit estimates for more than one asset or threat. Some studies did not assess costs/benefits by asset category, instead considering eradication or control costs associated with threats; 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 2 (262 internationally) exceeds the number of studies (117). A small subset of the total number of estimates (52) derived from Australian studies.

Our compilation is neither definitive nor exhaustive, so does not describe the entire body of literature and under-states the true availability of information. A more targeted search of studies within the field of health economics would, for example, almost certainly identify many more empirical studies that assess the impact of pathogens on human health. Nevertheless, this compilation clearly demonstrates that most empirical work has concentrated on a subset of threats and assets/cost categories (control costs and portfolio industries). Internationally, and in Australia, more than 50 % of estimates relate to control costs or portfolio damages (Table 2). 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 other asset categories. Most notable are the knowledge gaps relating to the potential 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 threats, but these impacts were not monetised.

3. Methods

There is insufficient knowledge for a comprehensive, whole-of-system assessment which relies on 'adding' individual impact estimates. Not only are the individual estimates generated using different methodological approaches and are thus not strictly additive; there are too many gaps. Noteworthy is the fact that studies which consider non-market values in general 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 and De Groot 2010),[1] for example, contains no fewer than 1310 value estimates, 116 of which are Australian. We thus chose to approach the problem by determining first, the value of the services flowing from different types of assets that are potentially 'at risk', from biodiversity threats (section 3.1). Using further insights form the literature, we then consider what might happen to the value of those service flows, if faced with different types of incursions (from different types of pests) – *vulnerabilities (*section 3.2). We spatially map both *values* and *vulnerabilities* clearly showing that various parts of Australia are more/less vulnerable to different types of biosecurity risks. We note that an important 'next step', which is not undertaken in this paper, is to incorporate estimates of *values* and *vulnerabilities* within a biological model that is able to properly model the spread of pests across space, and over time. That type of model would allow one to simulate (predicted) damages with and without various biosecurity interventions that would truly allow one to estimate the value of the biosecurity system (comparing future asset values with and without the system).

Although it is possible to simulate the spread of pests at fine geographic scale, the scale at which it is possible to estimate and thus spatially allocate the *values* associated with core assets 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 – and it is on these regions that we focus. Fig. 1 shows the boundaries of these regions with population densities, clearly highlighting the concentration of people around the coastline – particularly in and around the 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 knowledge to accurately estimate the *values* associated with human and social capital at 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. Details about the way in which we estimated the *values* associated with each asset category are provided below.

3.1. Estimating current values

3.1.1. Portfolio Services (Provisioning Services I)

3.1.1.1. Agriculture. The ABS provides data about the value of agricultural production, at NRM scale, for broad commodity groups (ABS, 2018). We worked with estimates for relatively aggregated commodity groups which could be linked to ABS production data to the two-digit land-use classification data from the Australian Land Use Mapping (ALUM) – thus generating spatially differentiated estimates for cropping, horticulture and livestock (intensive (e.g. pigs, poultry), and extensive (sheep cattle), derived by dividing the ABS estimates of production values, by the hectares of land used for that type of production, to generate NRM specific (per hectare) values. Variations in per-hectare values across NRMs thus 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. 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). See section 1, appendix A, supplementary materials for details and estimates.

3.1.1.2. Forestry. 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. Details are provided in section 2, appendix A, supplementary materials. We note that more sophisticated methods of estimating forestry values are multi-period and take into account both growth and harvest rates – the aim

[1] See also the Ecosystem Services Valuation Database (ESVD), which has expanded substantially since it's initial release based on the TEEB database https://www.esvd.info/.

Table 2

Count of empirical studies focused on the economics of biosecurity by asset and threat category. Total numbers shown in black; numbers relating to Australian studies and Australia studies that assessed 'consequence' shown in grey; asset/threat categories for which we could find no potentially transferrable estimates shaded in grey.

Asset Category	Threat category: Invertebrates - terrestrial	Vertebrates - terrestrial	Weeds - terrestrial	Pathogens (terrestrial and air)	Freshwater pests (all)	Marine pests (all)	Aquatic (marine or Fresh – 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									
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	5		3	7			1	2	18

being to estimate net present values (e.g. Creedy and 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.

3.1.2. *Non-Portfolio Services (Provisioning services II)*

Both Indigenous subsistence values and water values are examples of non-portfolio provisioning services. However, the supply-side approaches that are commonly used to estimate water values are inextricably linked to the methods used to estimate other water-related regulatory service values. Rather than explaining similar methodological approaches twice, we postpone discussing water values until section 2.4.3, focusing here, on Indigenous subsistence values.

With the deepest respect we acknowledge the breadth and complexity of Indigenous cultural values and connections to country, discussed widely in bespoke literatures. See, for example: Hill et al. (2011). We were fully cognisant of our inability to capture these values appropriately, so conferred with Indigenous scholars, seeking advice as to whether it was better to omit reference to Indigenous values altogether, or to flag their importance, albeit inadequately. We were encouraged to flag rather than ignore, thus do so in two ways: first considering only subsistence/food values; second more broadly considering other Indigenous cultural values (section 2.5.4.3).

Our estimates of Indigenous subsistence values use those reported in Sangha et al. (2019) as a base. This was an estimate of Indigenous subsistence food and material values for the Northern Territory: ≈ $500 per (Indigenous) person. This estimate is relevant to Indigenous people

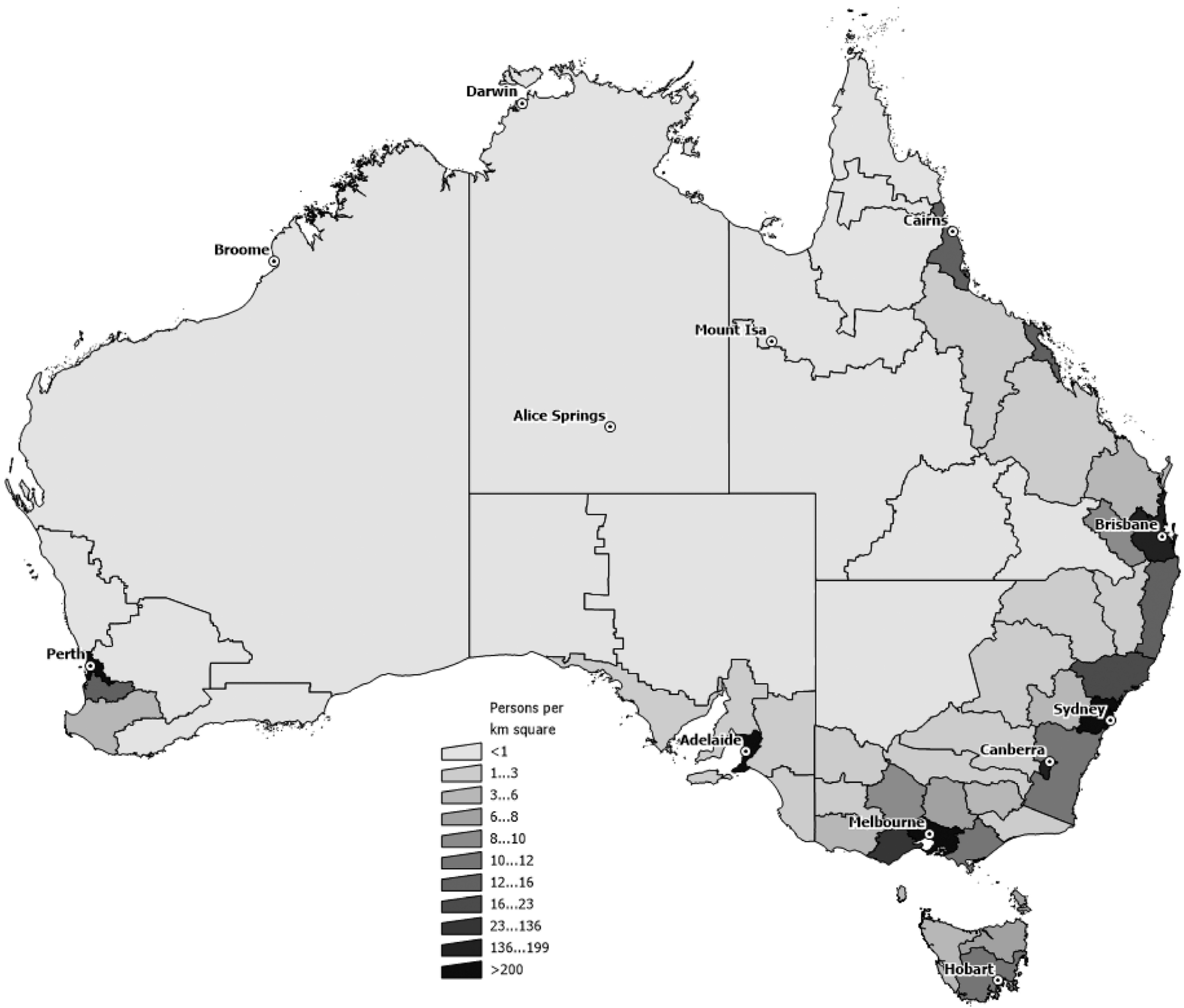

Fig. 1. The geographic resolution of our analysis: Natural Resource Management (NRM) regions and their population density.

who live '*On Country*' in remote areas but we feel it may be somewhat less relevant for Indigenous people living in large cities. We thus made adjustments to the base as below:

1) We calculated 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[2]: NRM^{Aria+} $(=\sum_{i-1}^{N}\frac{Areas_i \times ARIA+_i}{Area_{NRM}})$.
2) We estimated (per Indigenous person) subsistence food values within each NRM as a proportion of maximum values ($500) – that proportion calculated as: $\frac{NRMAria+}{15}$. The most remote regions (with Aria plus scores of 15) thus have food values of $500; in urban areas (with Aria plus scores of 0), subsistence food values are zero.

We estimated all-of-NRM Indigenous subsistence values by multiplying the Aria-weighted per-person values by estimates of the Indigenous population. Ontological differences between Indigenous and non-Indigenous perspectives, data deficiencies and imperfections in methods, ensure that our estimates are imprecise at best – and we urge readers to consider them as *place holders* rather than as figures that can be relied upon.

3.1.3. Water and other regulating services

First, we consulted Van der Ploeg and De Groot (2010)'s database, identifying studies that had generated estimates of water and other regulating service values. We supplemented that compilation through additional literature searches, used to identify more recent estimates. These searches focused primarily, although not exclusively on studies undertaken in Australia, and/or in arid regions (which comprise so much of the Australian landscape). We used sub-categories of ES from the CICES (Haines-Young and Potschin, 2012) to guide the search and compilation of data. 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 initially aimed to differentiate between water for drinking and sediment control but were only able to find two studies that unambiguously and exclusively focused on sediment – those that considered sediment often also considered water quality / water purification more broadly. We thus grouped all studies that referred to values associated with 'drinking water', 'water purification' and/or 'sediment' together under the heading 'water purification'.

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

[2] There were changes to NRM boundaries between 2011 and 2016, so relations are approximate only.

Vegetation Inventory System[3] so we could allocate values spatially (Supplementary Materials, Appendix B). We then used information from each individual study in our compilation of data, to determine which of our four 'biomes' each (sub) ES value estimate referred to.

Third, we estimated transfer functions for each (sub) ES/biome. We did this in recognition of the body of literature demonstrating that one should not simply transfer estimates from one region to another without 'contextualising'. There is evidence to suggest that one can reduce transfer errors by considering biophysical factors related to the type, quantity and 'quality' of biome (Tardieu and Tuffery 2019) – as we have done above, grouping estimates according to the (best matched) vegetation type.

It is also clear that socio-economic context matters, although there are no clear cut rules to determine the 'best' variables for use in transfer functions (Baker and Ruting, 2014). Variables capturing the socio-economic status of population are commonly used (Johnston et al., 2017) as are demographic (Paracchini et al., 2014), geospatial (Fitzpatrick et al., 2017) and 'perception' variables (Farr et al., 2016). We thus reviewed meta-analyses of studies undertaken across multiple countries to identify socio-economic factors which are commonly found to be statistically significant correlates of ES values.[4] Noting there is a difference between what should be done and what could be done, we chose to focus on just three commonly used variables (abundance of biome, population density, income per capita) which, for this study, were readily available for both study and transfer sites.

For study sites, we sourced original studies, compiling 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 to estimate those variables, we sought such information from other sources[5]; if unobtainable, the study was omitted. In some cases, a single study used almost identical approaches to estimate multiple values for different sub-biomes (e.g., 4 different types of grass). In these situations, we recorded a single (mean) estimate in our database – rather than entering each separately. To do otherwise, would be to, essentially, weight our pooled estimates in favour of a single study/approach. Abundance was estimated as area of study site divided by area of county. Our final compilation included 249 estimates relating to 4 biomes, and the six different (sub) regulatory service values – although there were only 208 cases without missing data (Table C1, Supplementary Materials, Appendix C lists all studies and provides descriptive statistics, by biome and type of regulating service).

We converted value estimates to 2015 AUD (per hectare, per annum) – estimating transfer functions by regressing logged estimates of per-hectare values against (logged) measures of abundance, per-capita GDP and the population density of the country in which the study had been undertaken. There were too few observations to estimate 24 separate transfer functions for each ES / biome, so we only estimated six: one for each type of (sub) regulating service value (Table 3). We used quantile regression to minimise the influence of outliers.[6]

There were no statistically significant coefficients for models 3 – 6 so we did not use coefficients from those models to estimate NRM-specific values for flood control, genepool/nursery values, carbon C-sequestration or the mediation of soil and air. Instead, we simply pooled estimates (Brouwer et al., 2015) multiplying the median value-per hectare by the total number of hectares within each NRM to estimate total NRM values, for each biome and service. There were two exceptions:

1) Studies specific to woodlands/savannahs and shrublands/grasslands were sparse, so those two biomes were grouped (hereafter, this larger group is termed drylands). There are, however, sound biophysical reasons for believing that the per-hectare 'value' of the services (strongly associated with vegetative cover) will be less in the former than the later. So, for all regulating service values we used one-half of the median value per hectare associated with the (grouped) biome - drylands – when estimating values for shrublands/grasslands.
2) When considering gene-pool/habitat values we also distinguished between freshwater and marine environments since there were numerous studies of both wetlands and mangroves – and it was thus possible to retain the benefits of 'pooling' without combining MVG 23 and 24. There were also a handful of studies that provided estimates of gene-pool values in estuaries (MVG 28).

For water purification and erosion, we used coefficients from the transfer functions to develop equations describing the relationship between per-hectare values, abundance, population density and per-capita income for each type of biome. We substituted NRM specific measures of those variables (Supplementary Materials, Appendix D) into the equations – thus providing some socio-economic contextualisation when transferring estimates. We also allowed for biophysical differences – replacing the constant/intercept term for each equation with the median (sub) ES value relevant to each biome. As previously, woodlands/savannahs were assigned and initial (intercept) term equal to the median for *drylands,* grassland/shrublands were assigned one half of the median for *drylands.*

Finally, we repeated the estimation process, using the per-hectare values associated with first, the lower (Q1) and second, the upper (Q3) quartiles – providing a somewhat crude, albeit important, sensitivity analysis. We did not do a similar sensitivity analysis around socioeconomic variables, since the transfer function coefficients were quite small (so a related analysis would not substantially affect the range from associated with our crude quartile test).

3.1.4. *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 an allowance for Indigenous cultural values.

3.1.4.1. 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

[3] The NVIS provides data on vegetation across all of Australia. See also: https://www.environment.gov.au/node/18927.

[4] Lara-Pulido, Guevara-Sanginés, and Martelo (2018); Chaikumbung, Doucouliagos, and Scarborough (2016); Carrasco et al. (2014); Salem and Mercer (2012); Brander et al. (2012); De Groot et al. (2012); Ghermandi et al. (2008); Brander, Florax, and Vermaat (2006); Reynaud and Lanzanova (2017); Wright and Eppink (2016); Siriwardena et al. (2016); Ghermandi and Nunes (2013); Brander, Van Beukering, and Cesar (2007); Schägner et al. (2018).

[5] Sourced from: https://data.worldbank.org/country/ (for all countries except French Polynesia – data sourced from https://www.fao.org/faostat/en/#data/MK). GDP per capita measured in 2010 US$, then converted to AUD using the World Bank PPP, and converted to 2015 AUD using Australia's GDP Deflator (also from the World Bank).

[6] 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).

Table 3
Quantile regression results from meta-analyses of water and other regulatory values - data collated from (Van der Ploeg and De Groot 2010) and elsewhere; p-values (in brackets) for robust standard errors.

	Model 1 Water purification	Model 2 Erosion	Model 3 Flood control[1]	Model 4 Genepool[2]	Model 5 C-sequestration	Model 6 Mediation
Ln (StudyArea / CountryArea)	−0.3178 (0.009)	0.0434 (0.902)	−0.2698 (0.253)	−0.0753 (0.505)	−0.1744 (0.195)	0.3606 (0.293)
LnPopDensity	0.8324 (0.011)	0.9833 (0.363)	0.3211 (0.655)	0.4430 (0.131)	0.2457 (0.530)	−0.2302 (0.605)
LnGDPPerCapita	0.1087 (0.675)	0.9101 (0.099)	−0.3271 (0.253)	−0.0210 (0.944)	−0.0689 (0.795)	−0.8271 (0.293)
Constant	−1.6665 (0.632)	−7.6835 (0.300)	4.4280 (0.427)	2.7794 (0.421)	2.5696 (0.364)	14.9564 (0.240)
N	45	20	37	68	28	10
Pseudo R-squared	0.3184	0.2001	0.1528	0.0667	0.0767	0.2190

[1] There were 30 separate estimates of flood control values in wetlands, so we tried a regression on that subset only but none of the coefficients were statistically significant.
[2] We also ran regressions investigating genepool estimates within particularbiomes for which we had relatively large sample sizes (23 Mangroves; 20 forests); none of those coefficients were statistically significant.

aesthetic, amenity and recreational values as if they capture a single 'use value', rather than treating as separable and adding.[7]

Here too, we first consulted Van der Ploeg and De Groot (2010)'s database, identifying studies that had generated estimates of use-values. That 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 focus on 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 too. 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.

With this sample of estimates (N = 136, see Supplementary Materials, Appendix E), we estimated transfer functions using quantile regressions (Table 4). We used coefficients from the transfer functions as parameters in equations describing the relationship between per-person values, abundance, population density and per-capita income for each type of biome. We substituted NRM specific measures of those variables (Supplementary Materials, Appendix D) into the equations – thus providing some socio-economic contextualisation to estimates. We also allowed for biophysical differences – replacing the constant/intercept term for each equation with the median (sub) ES value relevant to each biome. As previously, we divided the median estimate for *drylands* by two, using that as the base value for shrublands and grasslands. Per person use-values were assumed constant for beaches and wetlands since none of the coefficients in those transfer functions were statistically significant.

Use values are relevant to both residents and tourists, but we could not find publicly available data to estimate net domestic tourism at each NRM (tourism authorities focus on state of origin rather than NRMs). Even if such information were available, it would be a non-trivial exercise to derive net-tourism estimates from it to ensure no double-counting. We thus took the pragmatic step of estimating use values for residents and international tourists separately – noting that these estimates could be improved upon in future studies. Specifically, we:

1. Assumed that all resident use values are associated with the NRM in which a person lives. Total residential use values in each NRM were thus simply estimated by multiplying per person values by population estimates.
2. Used published data to estimate international tourist numbers for each NRM,[8] multiplying those estimates by per-person use values to generate estimates of total cultural-use values associated with international tourist.

3.1.4.2. Cultural services II: Non-use values. There were relatively few non-use value estimates in (Van der Ploeg and De Groot, 2010)'s database so we sourced most estimates from elsewhere – focusing mainly, although not exclusively, on research undertaken in Australia. As previously, we converted all estimates to AUD, 2015, 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/region in which the study site was located (done for regulating services and cultural use-values), 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 that GDP 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 value (N = 69, details in Supplementary Materials, Appendix F) to estimate 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) – Table 5. We used the coefficients as parameters, and data relevant to NRMs, as previously, to generate NRM-specific estimates of cultural non-use values.

Several researchers in Australia and elsewhere have considered the issue of *distance decay,* which describes the phenomena where people who live close to an environmental asset hold higher non-use values than people who live further away. There is mixed evidence of its existence in Australia: Rolfe and Windle (2012), for example, find evidence

[7] We do, however, acknowledge that some studies are designed to carefully separate aesthetic, amenity and/or recreational values and thus produce additively separable values.

[8] The boundaries of tourism regions for which data are available do not generally coincide with NRM regions, so when estimating tourism numbers to each NRM, we assumed that visitor numbers (reported by Tourism Region (see Table S3.3, Appendix S3) were scattered equally across space.

Table 4
Quantile regression results from meta-analyses of cultural use-values - data collated from (Van der Ploeg and De Groot 2010) and elsewhere; p-values (in brackets) for robust standard errors.

Dependent variable = Ln (Value per Person)	Model A Terrestrial (excludes coastal)	Model B Drylands (excludes wetlands)	Model C Forests	Model D Wetlands	Model E Coastal (beach recreation)[1]
LnPopDensity	−0.6719 (0.001)	−0.8008 (0.019)	−0.6664 (0.029)	−0.4912 (0.104)	0.0657 (0.993)
LnGDPPerCapita	0.0294 (0.910)	0.0033 (0.992)	0.0203 (0.950)	−0.1405 (0.611)	0.2318 (0.994)
Ln(StudySiteArea /Country Area)	0.0915 (0.147)	0.0980 (0.263)	0.0843 (0.319)	0.1756 (0.208)	−0.1402 (0.791)
Constant	7.7800 (0.003)	8.0439 (0.066)	7.7507 (0.063)	8.8914 (0.023)	0.6417 (0.999)
N	47	27	22	18	22
Pseudo R-squared	0.2117	0.2292	0.2572	0.2619	0.0263

[1] These values are unaffected by terrestrial pests but we have nevertheless retained them here for completeness.

Table 5
Quantile regression results from meta-analyses of cultural non-use values; p-values (in brackets) for robust standard errors.

Models with dependent variable = Ln (Value per Hectare)	Model ATerrestrial (excludes marine/reef)	Model BDrylands (excludes wetlands and marine/reef)	Model C Forests	Model D Wetlands
LnPopDensity	0.3885 (0.694)	4.5405 (0.003)	3.6184 (0.028)	1.5951 (0.054)
LnGDPPerCapita	−0.0561 (0.960)	−0.8217 (0.363)	−2.0711 (0.114)	1.7539 (0.169)
Ln(StudySiteArea /Country Area)	−0.3601 (0.195)	−0.4045 (0.007)	−0.2773 (0.096)	−0.3161 (0.033)
Constant	−0.2054 (0.988)	−9.9464 (0.458)	7.2758 (0.671)	−20.8333 (0.163)
N	29	19	13	10
Pseudo R-squared	0.1593	0.4900	0.5914	0.2761

that residents of Queensland have higher non-use values for the Great Barrier Reef than people who live elsewhere – although they find little to no evidence of distance decay in their 2003 study of non-use values in the Fitzroy estuary (Rolfe and Windle, 2003). Morrison and Bennett (2004) find that some non-use values for rivers in South-Eastern Australia are lower amongst people who live close to the rivers than for people who live further away,[9] but Zander et al. (2013)'s research suggests that distance decay may be present for rivers in Northern Australia. Evidently, the presence/absence of distance decay is context specific. We do not have enough information to appropriately test for, and where necessary, nuance estimates to allow for distance decay across all biomes and NRMs – so we instead assume it is absent, noting this issue as a potentially important one for future research.

3.1.4.3. Cultural Services III: Placeholder 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 they are rarely quantified or valued in monetary terms. For Indigenous people, going out 'on country' generates documented improvements in mental and physical health and in social wellbeing (Burgess et al. (2005), Garnett et al. (2009)) – similar to the physical and mental health impacts that have also been observed for non-Indigenous urban dwellers who are able to spend time 'with nature' (Bratman et al., 2012). Being out on country also provides Indigenous people with opportunities to gather food and bush medicines for personal consumption, ceremony and tradition. Eco-enterprises associated with land and sea management activities create employment for the Indigenous people and active involvement in other industry sectors such as mining and tourism, help reduce welfare dependence (Zander and Garnett 2011). Healthy ecosystems are known to contribute to education outcomes with visits 'on country' providing appropriate forums for cultural knowledge exchange (Casimirri, 2003; ALSC, 2004; Abah et al., 2015; ANKN 2008; Ammann et al., 2007). Moreover, there are significant ties to country – even those who live far from their traditional lands, report measurable increases in wellbeing from knowing that their country is being looked after (Larson et al., 2018). The relationship between Indigenous people and country is thus complex, inter-connected, reciprocal and inherently different from the use and non-use cultural values discussed above (Stoeckl et al., 2021).

Noted earlier, we conferred with several Indigenous scholars who gave in principle support to the idea of including very rough estimates of Indigenous cultural values in this broader assessment, if only, because to exclude them altogether may be to risk implicitly assigning them a value of zero. There was, however, collective agreement on the need to ensure that more detailed work is undertaken to properly assess these values, and agreement that this work can only, rightfully, be led by Indigenous scholars. The values we provide here can thus only be considered as a placeholder. Studies that we use to provide these estimates are briefly described below:

1) Some studies have sought to generate monetary estimates of the value of Indigenous cultural connections to country by considering benefits derived therein, and then determining how much it would cost to achieve similar outcomes (e.g. health, education, self-confidence) using formal western methods of doing so. Sangha et al. (2019) estimate those benefits to be approximately $5,500 per person, per annum.
2) Zander and Straton (2010) report on a choice-modelling study, undertaken in three river catchments across Australia's north. Respondents were asked to express preferences for different development options that would impact areas in and around the rivers. Despite having significantly lower incomes, and facing significantly higher living costs (e.g., prices in local stores), the welfare impacts of changes to the natural environment were much larger for Aboriginal than non-Aboriginal respondents. Indigenous people were, on average, willing to pay between $252 and $1752, per person, for changes that improved the local condition of water holes from 'poor' to 'good' – suggesting that the holistic range of benefits obtained from (improved) water holes exceed, on average, $1002 per person, per average.
3) There have been several assessments of the *social return on investments* in Indigenous land and sea management programs

[9] These were implicit prices of 'healthy vegetation' and 'native fish species' in the Gwydir and Murrumbidgee river catchments; distance decay was, in contrast, apparent for use-values in those catchments (associated with the implicit prices for swimming and fishing).

undertaken by Social Ventures Australia (SVA) in communities across Australia, with an average per-person estimates of the social, economic, and cultural value of these programs being $3139 per annum. These studies include:

- o Kanyirninpa Jukurrpa's (KJ) on-country programs in Western Australia (Social Ventures Australia Consulting, 2014), estimated to have generated returns for community members (beyond monies paid as salaries), valued at approximately $3867 per person per annum ($1774 per person per annum in cultural values[10] and $2133 per person per annum[11] in social and economic values).
- o The Warddeken Indigenous Protected Area (IPA) and associated Indigenous Ranger programs in the Northern Territory (Social Ventures Australia Consulting, 2016b), estimated to have generated returns for community members which amount to approximately $2,648 per person per annum (between $994 per person per annum for rangers[12] and $1654 per person per annum[13] for the wider community).
- o Girrigin's IPA and associated ranger programs in QLD (Social Ventures Australia Consulting, 2016a), estimated to have generated returns for community members which amount to approximately $2902 per person per annum ($1105 per person per annum for rangers[14] and $1797 per person per annum[15] for the wider community).

We use a cultural-value-estimate of $3100 per person per annum – the median of (1), (2) and (3). Multiplying $3100 by the estimated number of Indigenous people living on mainland Australia gives a collective placeholder 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 the goods/services used in these bench-mark valuation studies.

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. Indigenous cultural values may thus best be considered place-based, and thus perhaps best 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.

3.1.5. Companion animals

Following the lead of O'Sullivan (2012), we attempted to consider different groups of animals: (1) those kept primarily as household pets (including horses for 'recreational/social' purposes and animals such as working farm dogs); and (2) those associated with the racing industry (greyhound dogs and horses). We could not, however, find data on (2) which could be reliably allocated across space, so focused on animals kept as pets. To the best of our knowledge, no Australian researcher has generated pure welfare estimates for these animals, so we used measures of 'value' that are comparable to those used for portfolio industries – namely expenditures by owners of domesticated animals.

For cats, dogs and other small domestics animals, we used data from Animal_Medicines_Australia (2016) which reports average expenditure on pets per (owner) household (Table 6), 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) (Table 7). This allows us to allocate values across the landscape using ABS census data on usual place of residence. We note that the observed spatial variation in expenditure per capita is entirely based on observed differences in the average number of pets per household, with expenditure per pet assumed constant across Australia. Animal_Medicines_Australia (2016) notes the small sample size in the NT, recommending that those estimates be treated with some caution. Rather than using their reported NT estimates (which collectively add to $710 p.a.), we thus used the Australian mean values for residents of that territory).

We used 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 Australian's own (recreational) horses. Unable to find better information regarding the location of recreational horses, we assumed a constant rate of horse ownership across the entire country. Macleay (2018) found that more than 70 % of owners kept their horses within 5kms of their place of residence (8.9 % had horses 6–10 km; the remaining with horses more distant) – in the absence of better information, we thus allocated values spatially according to the distribution of population.

Gordon (2001) estimates that the annual costs of up-keep for recreational horses (which include agistment /stabling, feed, veterinary, dentist and farrier) are approximately $3000 in rural areas and $11,000 in metropolitan areas; the higher urban costs reflecting the need for stabling (converted to 2015 AUD). She also notes the significant expenditure differentials between what Gordon terms low-cost '*paddock bashers*' and high-cost '*serious event city horses*' (with all manner of horse, and expenditure, between). So we used Gordon (2001)'s estimates of costs which relate to horses kept for recreational purpose only: $8774 in metropolitan areas (converted to 2015 AUD); and $1500 for horses kept permanently in pasture. Our estimates thus understate the true costs of keeping a '*serious event city horse*' but are nevertheless consistent with those of the RSPCA[16] who suggest that it costs about $8400 per annum to keep a horse. Associated feed costs are also consistent with Macleay (2018)'s estimates of feed costs (she did not include other expenses) – reported at, on average, $2280 per horse, per annum.[17] Our exclusion of 'serious event' city horses, is also consistent with results from Macleay (2018)'s survey of horse owners, who reports that only 2.1 % of the horses covered by her study were permanently stabled.

When attempting to allow for different cost-structures in urban and rural areas, we used the ARIA + scores to provide a nuanced, sliding-scale estimate of the cost-per horse. This effectively assumes that expenditure falls by 5.53 % for each one-point increase in ARIA+ (with a total fall in expenditures from 8774, for ARIA+=0 to 1500 for ARIA+=15):

$$Value\ per\ horse\ in\ NRM_i = \$8774 \times (1 - ARIA + score\ in\ NRM_i \times 0.055)$$

3.1.6. Physical infrastructure

We rely primarily on the ABS experimental estimates of the value of

[10] Calculated by dividing their total estimate of cultural values ($17,337,000) by 5 (years relevant to the study) by 2000 (estimated population of the community).

[11] $21,332,000 divided by 5 (years) divided by 2000 (population).

[12] $7,159,039 divided by 6 (years) divided by estimated population (1200).

[13] $11,914, 922 divided by 6 (years) divided by estimated population (1200).

[14] $4,642,219 divided by 6 (years) divided by estimated community population (700).

[15] $7,551,437 divided by 6 (years) divided by estimated community population (700).

[16] "Farriery every 6–8 weeks = $50-$80 per month, feeding costs (when not on grass alone)= $200-$400 per month, worming every 6–8 weeks =$15 per month, veterinary care (vaccines / wound care etc) =$100 per month, annual dentistry= $10 per month, agistment fees =$120–300 per month" – quoted from RSPCA website, https://kb.rspca.org.au/knowledge-base/when-is-the-right-time-to-buy-a-horse-pony-for-my-child/.

[17] These numbers are slightly lower than estimates from a report from the horse industry council, which surveyed more than 3000 horse owners (only 6% of which were racers) noting that they collectively spent about $40m during the 2014/15 financial year – about $13,000 per respondent per annum https://www.horsecouncil.org.au/wp-content/uploads/2015/08/2014–15.jpg.

Table 6
Pet ownership and average expenditure across all of Australia, 2016.

Animal	Estimated number in Australia	Average per (owner) household	Estimated number per 100 people	Households with these animals ('000)	Estimated expenditure per (owner) household
Dogs	4,759.7	1.3	20	3,600	1,975
Cats	3,883.6	1.4	16	2,700	1,480
Fish	8,729.5	8	37	1,100	403
Birds	4,187.4	3.9	18	1,100	444
Small mammals	536.9	1.9	2	283	464
Reptiles	415.5	1.7	2	250	633

Table 7
Pet ownership – % of households with at least one animal, and expected value of expenditure per capita (estimated from data), by state/territory, 2016.

Animal		Australia	NSW	ACT	VIC	TAS	QLD	SA	WA	NT
Dogs	% of households with at least one animal	38	38	43	40	43	37	45	33	45
	Expenditure per capita (expected value)	289	289	327	304	327	281	342	251	289
Cats	% of households with at least one animal	29	25	34	34	30	26	37	28	45
	Expenditure per capita	165	142	194	194	171	148	211	159	165
Fish	% of households with at least one animal	12	15	16	11	14	9	9	9	18
	Expenditure per capita	19	23	25	17	22	14	14	14	19
Birds	% of households with at least one animal	12	14	16	10	7	10	13	12	27
	Expenditure per capita	20	24	27	17	12	17	22	20	20
Small mammals	% of households with at least one animal	3	3	2	4	5	2	4	3	9
	Expenditure per capita	5	5	4	7	9	4	7	5	5
Reptiles	% of households with at least one animal	3	3	0	3	0	2	4	1	9
	Expenditure per capita	7	7		7		5	10	2	7
Other pets	% of households with at least one animal	3	2	7	4	5	2	3	4	0
Any pet	% of households with at least one animal	62	60	75	65	66	59	68	57	82
Sum of all expenditure per capita		505	490	577	546	541	469	606	451	505

capital stock, for each state (ABS 2017). These estimates provide information on: gross fixed capital formation, end of year net capital stock and the consumption of fixed capital, for numerous different categories of equipment/infrastructure and for different industries. We used estimates of net capital values, focusing on just a subset of all types of capital – specifically those relating to the electricity gas and water industry, and the housing stock (dwellings). We did this because the international literature suggests that these are the types of infrastructure most susceptible to pests. There is, for example, much evidence of the damage which Zebra mussels have caused to infrastructures associated with power and water generation (see, for example: Lovell, Stone, and Fernandez (2006) and Arthur, Summerson, and Mazur (2015)). Terrestrial pests are also a threat to infrastructure that is essential to the generation and transport of electricity (K.M. Burnett et al., 2008) and to people's homes.

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: other equipment (e.g. computing, electronics, motor vehicles) is unlikely to be susceptible. We thus estimate that the value of utilities 'at risk' (to pests) is 24 %

Table 8
Net capital stock for electricity, gas, water and waste (ALUM 5.6): total value, value 'at risk' and value per hectare, by state.

State	Population Persons	Capital Type	Net capital stock ($m)	Capital stock 'at risk'		
				Total value* ($m)	Annualised value# ($m p.a.)	Annualised value per capita ($ p.a.)
NSW	7,467,920	Utilities	97,681	23,443	1,196.04	$160.16
		Housing	549,487	131,877	6,728.267	$900.96
VIC	5,919,036	Utilities	97,879	23,491	1,198.49	$202.48
		Housing	460,529	110,527	5,639.006	$952.69
QLD	4,690,525	Utilities	63,375	15,210	776.00	$165.44
		Housing	363,092	87,142	4,445.920	$947.85
SA	1,673,783	Utilities	26,746	6,419	327.49	$195.66
		Housing	108,733	26,096	1,331.399	$795.44
WA	2,468,022	Utilities	35,993	8,638	440.70	$178.57
		Housing	208,514	50,043	2,553.157	$1,034.50
TAS	508,981	Utilities	8,790	2,110	107.65	$211.50
		Housing	32,096	7,703	393.001	$772.13
NT	226,276	Utilities	4,131	991	50.56	$223.44
		Housing	18,739	4,497	229.434	$1,013.95
ACT	396,853	Utilities	4,537	1,089	55.56	$140.00
		Housing	34,789	8,349	425.960	$1,073.34
TOTAL	23,351,396	Utilities	339,132	81,392	4,152.49	
		Housing	1,775,979	1,775,979	1,775,979	

* 25% of Net capital stock.

\# Assumes a 3 % discount rate and a 30 year time horizon: $= \frac{0.03*(\text{Total value})}{1 - 1.03^{-30}}$.

of the net capital stock for utilities. We generated NRM-specific estimates (Table 8) in several related steps. First, we annualised estimates, and divided through by state population, to derive an estimate of the average per-capita value of utilities 'at risk', for each state. We then exploited 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):

$$\frac{value\ of\ utilities\ at\ risk\ in\ NRM_i}{hectare} = \frac{State\ capital\ at\ risk}{State\ Population} \times \frac{Population\ of\ NRM_i}{Hectares\ of\ Utilities\ in\ NRM_i}$$

We then generate estimates of the total value of utilities 'at risk' in each NRM:

$$= \frac{value\ of\ utilities\ at\ risk\ in\ NRM_i}{hectare} \times Hectares\ of\ utilities\ n\ NRM_i$$

We followed a similar procedure for dwellings – noting that not all of the housing stock is susceptible to damage from pests (e.g., there are many brick homes with metal frames). 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 et al. (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.[18] In the absence of other information, we tentatively suggest that it may be appropriate to use the same % here (24 %) as for utilities. We thus estimated the value of the dwelling stock 'at risk' at 24 % of the net capital stock. As previously, we annualised the total stock values and then divided by (state) population to estimate the average per-capita value of dwellings 'at risk', for each state.

3.2. Vulnerabilities

3.2.1. Background information about the (potential) impact of pests on various ES

First, we focused on portfolio assets. The middle column of Table 9 provides a summary of estimates generated from studies that have reported on the impact that pests have had on Agriculture (see Table 1 in Appendix G for a full list of studies used in this analysis). These estimates can be crudely translated into estimates of the likely percentage loss in agricultural production by simply dividing the estimated % loss in GDP, by estimates of the % contribution that Agriculture makes to GDP (2.5 % in Australia) – see Table 9.

Next, we focused on studies that report pest damages (or WTP to avoid damages) to other ecosystem services. We categorised each estimate according to both the type of pest considered (often more than one), and the type of asset considered (often more than one). Articles that reported damage estimates from multiple pests or multiple asset groups without differentiating were excluded (since it was not possible to attribute damages from specific pests to specific assets). In all cases, damage estimates were converted to AUD 2015 (consistent with previous estimates). If a study reported estimates of both damages and existing (total) service-value flows, we used these bespoke estimates to calculate damages as a percent of existing service value flows.[19] If a study only reported damage estimates, we divided the reported damage estimates (expressed in $ per hectare, or $ person, depending on type of asset) by matched estimates of service values, from section 3.1. Matching was done, by pest group, asset type and biome. Estimates that could not be matched were omitted. Summary statistics are provided in Table 10[20].

There are relatively few studies overall, with significant knowledge gaps and widely differing estimates for some pest/service combinations. This was not unexpected (see discussion from section 2). Most notable, is the absence of any data that allows us to estimate damages as a percent

Table 9
Estimates of expected damages to portfolio industries.

Pest group*	Estimated % reduction in the value of GDP if incursion (Median)	Inferred % loss in value of portfolio service flows, assuming Agriculture contributes to 2.5 % of GDP
Invertebrates	0.7432	29.73
Vertebrates	0.1004	4.02
Weeds	0.2333	9.33
Pathogens (all)	0.0003	0.01

*None of the studies in Appendix G specifically considered the impact of aquatic pests on portfolio industries, so we have omitted that pest group from this analysis.

Table 10
Summary statistics relevant to studies from which it was possible to derive a 'reasonable' estimate of the potential damages that groups of pests are likely to impose on current assets (expressed as a percent of current values).

	Pest Group				
Asset	Invertebrates	Vertebrates	Weeds	Pathogens*	Aquatic
Studies reporting estimates specific to **Subsistence: NONE**					
Studies reporting estimates specific to **Water and other Regulating Services**					
N			8		
Mean			90.30 %		
Median			6.15 %		
Min			−11.46 %		
Max			659.47 %		
Studies reporting estimates specific to **Cultural Services**					
N	11	3	12		4
Mean	13.08 %	15.31 %	15.65 %		0.42 %
Median	11.75 %	7.33 %	2.55 %		0.42 %
Min	2.78 %	0.16 %	0.84 %		0.11 %
Max	30.91 %	38.44 %	125.47 %		0.72 %
Studies reporting estimates specific to **Infrastructure**					
N	4	1			
Mean	3.38 %	3.11 %			
Median	1.11 %	3.11 %			
Min	0.03 %	3.11 %			
Max	11.27 %	3.11 %			
Studies reporting estimates specific to **Companion animals: NONE**					

[18] The expected value of damages depends on the probability of incursions, and will thus be less that the value of stock 'at risk'. Guillebeau, Hinkle, and Roberts (2008) also report that in the U.S, pest management firms earned approximately $12.58 billion (AUD, 2015) in revenue from residential (general insect control and termites) and commercial services – we estimate this at approximately 0.03% of the value of the housing stock; in New Orleans termites are annually responsible for an estimated $100 million in damage to homes and businesses in the – approximately 0.26% of the value of the housing stock (a cursory google search => Population of New Orleans about 400,000 – average household size 2.6 so estimated number of dwellings = 150,000. Median home price in New Orleans placed at $176,000 ($257,000 AUD, 2015) so a very rough estimate of the value of housing stock = 150,000 * 257,000 = 38,550 million. If 100 million damage each year then suggests 0.26% value of housing stock damaged each year). Collective value of all US homes estimated at $46,489b (AUD 2015) in late 2017. https://zillow.mediaroom.com/2017–12-28-All-U-S-Homes-Worth-Cumulative-31–8-Trillion.

[19] If damage estimates were reported as $ per-person, we divided these per-person damage estimates by matching per person current value estimates, to estimate damages as a percent of current values for each pest/ES pair. If damage estimates were reported as per hectare, we divided per-hectare damage estimates by matching per hectare current value estimates, to estimate damages as a percent of current values, for each pest/ES pair.

[20] Table 2, Appendix G provides a full list of studies used in this analysis.

of *values* for subsistence food values or companion animals, for any pest group. The only studies that consider the impact of pests on regulating services have been those that consider weeds. The most comprehensively covered asset category relates to Cultural services although 12 of the individual estimates derive from a single study (Akter et al. (2015)'s investigation of people's WTP to reduce three different types of impacts, in three different Australian states (and pooled across all three states)). Other notable information deficits are associated with damage estimates relevant to Pathogens (N = 0) and Vertebrates (N = 4).

3.2.2. Generating regional estimates of vulnerabilities

Real world incursions play out dynamically across space and over time – as when a new pest arrives at an entry port, and slowly spreads across the landscape. It is thus a non-trivial task to estimate bespoke incursion costs (discussed in sections 1and 2) and generally requires the use of bio-economic models that allow for spatio-temporal dynamics. It is well beyond the scope of this article to do so, so we instead estimate what we term *vulnerabilities* – the potential losses that could ensue within a particular region, for different types of pests. We stress that these potential damage estimates (*vulnerabilities)*, do not provide good-quality information about potential damages in a particular location at a particular point in time. It is possible, for example, for a pest to be present across, say, 10 percent of a region. Damages in that small, affected, area may be very high (perhaps even choking out all service-value flows), but damages across an entire NRM may be comparatively small. Similarly, these estimates do not allow for the (very likely) situation of multiple, interacting, pests.

We focus on just two different types of incursions and three different asset types. These are the groups about which we have most information. For each NRM, we estimate simple vulnerabilities using only the estimates which we believe to be 'reasonably' robust, by adding the potential damages that a particular pest-type might cause to each asset – formerly, the potential reductions in service value flows that might occur. They are estimated as the % loss in service value flow (from Table 11) × existing service value flow (from Appendix F).

We present (estimated) *vulnerabilities* in two ways: as the potential loss in service-value flows per hectare; and as the potential loss of service-value flows across entire NRMs.

4. Results

NRM-specific estimates of at risk values are provided in the Supplementary Materials, Appendix H; Table 12 collates estimates across all of Australia – also reporting estimates that directly relate to the lower and upper quartile of estimates from studies used in the transfer functions (a somewhat crude, but nonetheless effective means of showing sensitivity and plausible ranges).

Even having omitted two crucially important assets (human health and social capital), our estimates of *values* add to approximately $250b per annum with a range of $174b to $1365b. Less than one-half of *values* are closely associated with the market (including those associated with portfolio assets, infrastructure and expenditure on companion animals); almost 60 % 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, and explicit price. This is in line with other research (see, for example, Costanza et al. (1997) and Wei et al. (2018)), and highlights the importance of considering both market (particularly portfolio industries) and non-market values for whole-of system assessments.

Table 11
Potential damages (as a percent of total value), by asset and Pest group.

	Potential damages, as a percent of total value, that could be incurred	
Asset	Invertebrates	Weeds
Provisioning Services (Portfolio)*	29.73	9.33
Water and other regulating services#		6.15
Cultural Services#	11.75	2.55

*Estimates from Table 9.
Estimates are Median values, from Table 10.

Fig. 2 shows the proportion of total *values* attributed to broad categories of assets by NRM, providing much evidence of spatial heterogeneity. In accordance with intuition, the *values* that are most closely associated with the market (shown in the left panels of Fig. 2) are much more important (when considered as a percent of total values) to NRM regions in the southeast and southwest than elsewhere in Australia. In other words, the high market-*values* are most important in NRM regions that have high population densities (shown in Fig. 1). Cultural service *values* are also comparatively important in NRMs with relatively dense populations (again, unsurprising, since in economics, cultural values are, by construct, per-person values). The non-market *values* that are most strongly associated with the environment (regulating services, water and subsistence values shown in the top two right maps of Fig. 2) are relatively more important in sparsely populated NRM regions than elsewhere.

The importance of that heterogeneity is further evidenced in Fig. 3 which shows *vulnerabilities* to weeds (left panel) and invertebrates (right panel). The data underpinning these maps is in Appendix I of the supplementary materials. The maps clearly show that *vulnerabilities* – whether measured as the potential losses per hectare (top panel) or potential losses per NRM (bottom panel) – are generally highest in the more densely populated areas (for both invertebrates and weeds). Vulnerabilities are also evident in some of the NRMs surrounding urban areas – regions with significant agricultural activity supporting adjacent populations. Vulnerabilities also tend to be greater for invertebrates than for weeds, although this is not true in many of the large, remote, and generally drier NRMs across the north, in the outback and in the west.

5. Discussion

Data deficiencies prevent us from undertaking a comprehensive assessment of the value of Australia's biosecurity system using empirical data and/or studies across the entire continent. So we, instead, use observable market prices and a benefit transfer approach to shed light on the issue – estimating the *value* of the services associated with the assets that are protected by the system. Benefit transfer is commonly used in the biosecurity literature, with researchers regularly compiling estimates (of likely losses in productivity, or costs of control) generated in other studies, for use in theirs. See for example: (K. Burnett et al., 2006; K. Burnett et al., 2007; K.M. Burnett et al., 2008) in their studies of the impacts of weeds and brown tree-snakes; Wylie and Janssen-May (2017)'s study of the potential 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. That said, most studies use simple value-transfers and focus on just one pest (or type of pest) at a time. Our study differs from these, in that we use transfer functions and consider a broad range of assets that are at risk from biosecurity threats. We do not consider impacts on human health and/or on social capital in this paper but flag them as important asset-values to be incorporated in future research.

We find that the biosecurity system helps protect assets which generate a flow of benefits valued at more than $250b p.a. and the majority of these values are ones not normally captured through the market (e.g., they relate to regulating and non-use cultural services). We are unaware of any other study that has sought to estimate the value of service flows for such a comprehensive 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

Table 12
Values that flow from the assets that are protected by Australia's biosecurity system $b, p.a.

Relevant Capital	Asset Broad class	Sub-class	Additional information	Estimated values ($m p.a.) "Best"	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	PLACEHOLDER	$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
Natural	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	*Omitted – (data deficiencies and exceedingly difficult modelling challenges)*			
		Domesticated animals (excluding horses)		$11,723	$11,723	$11,723
		Horses for recreation		$3,525	$3,525	$3,52a5
		TOTAL		$15,247	$15,247	$15,247
Physical	Infrastructure	Dwellings	24 % of net capital stock	$21,746	$21,746	$21,746
		Utilities (electricity)	(annualised)	$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

their earlier 1997 estimate of $1109 per hectare (Costanza et al., 1997). At least some of the difference between our estimates and those of Costanza et al (1997, 2014) are likely attributable to the fact that so much of Australia is arid and has low population densities, compared to the rest of the world. Our decision to estimate regulating service values for different vegetation types and to use transfer functions which adjust for GDP and population densities thus explain some of the differences. But our low estimates also reflect a deliberate research strategy. Whenever we had to make a methodological choice affecting value estimates, we selected the low value option. Although there is significant uncertainty in our estimates, we can, at least, be confident that our estimates are reliably downward biased, and defensible.

Despite the fact that more than 50 % of the studies included in our biosecurity research stocktake (Table 1) considered portfolio assets (those associated with agriculture, forestry and fishing) – these assets accounted for less than 25 % of all *values* 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 many do not 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 and De Groot, 2010).

6. Conclusion

Real world incursions play out dynamically across space and over time – as when a new pest arrives at an entry port, and slowly spreads across the landscape. It is thus a non-trivial task to estimate bespoke incursion costs and generally requires the use of bio-economic models that allow for spatio-temporal dynamics. It is an even less trivial task to estimate the entire value of the biosecurity system[21] and was well beyond the scope of this article to do so. We instead, estimated the value of the services provided by assets that are protected by the biosecurity system: ≈ $250b p.a., 90 % of which are generated from Australia's *Natural Capital.* We also estimated what we term *vulnerabilities* – the potential losses (to the estimated *values)* that could ensue within a particular region, for different types of pests.

Our NRM scale *value* estimates suggest that different regions are likely to be differentially *vulnerable* to different types of hazards. Areas inland from the major population centres derive a large proportion of their total values from portfolio assets (mostly, agriculture and forestry) and it is these regions which are at most *risk* to incursions that affect portfolio assets. The more populated regions near Australia's major cities derive most of their *values* from infrastructure, companion animals, and cultural use values (aesthetics, amenity and/or recreational values) – consequently, those regions are arguably at most *risk* to

[21] That would require one to estimate the value of all service-flows with and without a biosecurity system, for different pest arrival and dispersion scenarios, accounting for particular biosecurity measures, their costs, and their impact on arrivals and dispersions.

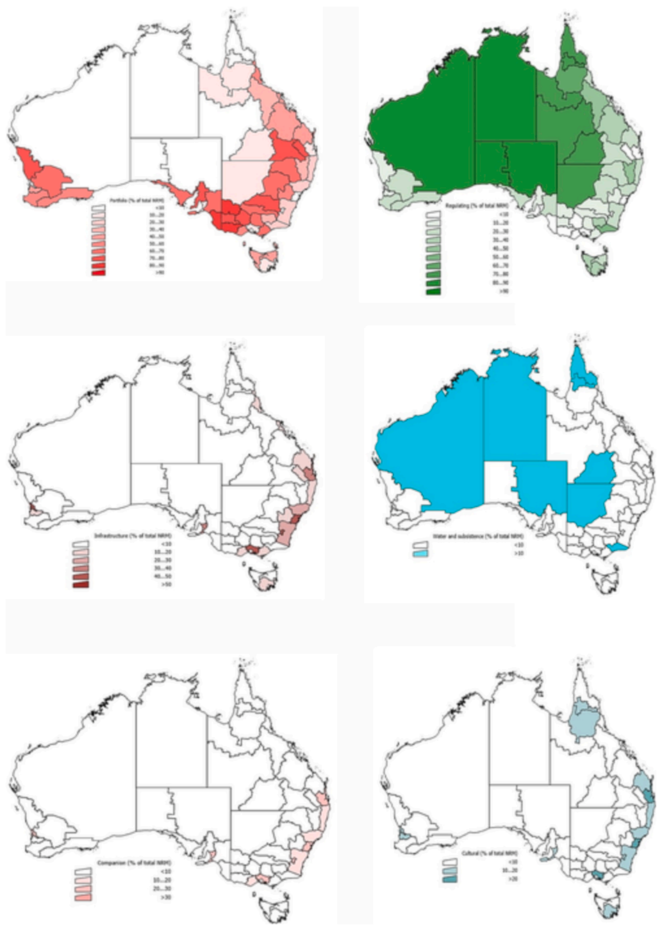

Fig. 2. Portfolio, infrastructure, companion animal, regulating, water & subsistence, and cultural service Values as a percent of total service values, by NRM (excluding human and social capital) – $m, p.a. Left column (assets closely related to the market): Top, Income from portfolio industries; Middle, Annualised value of 'at risk' Infrastructure; Bottom, expenditure on companion animals. Right column (assets not closely related to the market): Top, Regulating service values; Middle, Cultural values; Bottom, Water and subsistence values.

incursions that affect those associated assets (e.g. termites impacting infrastructure, Hendra-type viruses affecting horses, biting insects affecting the quality of outdoor recreational experiences – Akter et al. (2015)). Inland areas have a large proportion of *values* 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 particularly evident when considering subclasses of assets. Carbon sequestration values, for example, are driven by vegetation type with the highest values in mangroves, wetlands, forests and woodland areas. Regional variations in those values are thus driven by regional differences in vegetation. Similarly contrasting patterns can be found in each of the other asset classes and 56 NRM regions, suggesting that the realised impact of pests/diseases, or any other type of threat, is likely highly dependent on when and where an organism establishes (or threat impacts) in the first instance.

Our relatively simple scenario analysis showed that regional *vulnerabilities* to both invertebrates and weeds – whether measured as the potential losses per hectare or potential losses per NRM – are generally highest in populated areas. Many regions are more *vulnerable t*o invertebrates than for weeds – particularly some of the NRMs surrounding urban areas with significant agricultural value. However, the large, remote, and generally dry NRMs across Northern Australia, in the

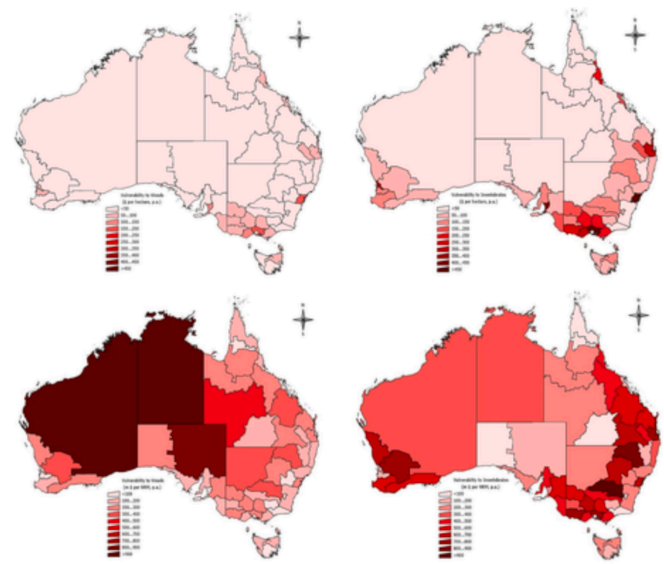

Fig. 3. Regional vulnerabilities to weeds (left) and invertebrates (right), by NRM. Top panels show potential losses as a percent of the total value of asset flows; Bottom panels show total value of potential losses ($m, p.a.), by NRM values as a percent of total assets (excluding human and social capital).

outback and in the west were more *vulnerable* to weeds than to invertebrates. Critically, we know very little about the damage that pests and disease cause to regulating services (the most important service value in those NRMs) since most biosecurity literature focuses on the potential damage of incursions on market values (e.g. on agriculture, forestry and physical infrastructure).

We stress that our estimates do not tell us how much the biosecurity system is worth; neither do they tell us how much damage a particular pest, landing at one place, at one point in time, will inflict upon a region. But our spatially explicit estimates could be embedded within a larger bioeconomic model to simulate the spread (and monetary impact) of various pests across space and over time – perhaps even with and without various biosecurity measures. Large and ambitious, such a model could – in theory – generate estimates of the value of the system as a whole. Developing such a large and ambitious model is an important end-goal. When assessing values and impacts 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 pests or assets could unintentionally focus thought, policy and resources on pests/assets which are easy to value – they may not necessarily be the pests/assets that are most important. Data deficiencies generally prevent large-scale assessments, but our work demonstrates a novel way of circumventing that problem, and (as above), lays solid foundations for an improved whole-of systems model. Although imperfect, our work thus represents a significant step forward, greatly improving the information available to Australian policy makers.

There is a broader benefit from adopting an asset led framework such as the one presented in this paper – 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. 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. Our approach, and our estimates, thus present an opportunity for economies of scale. Non-market valuation is frequently cited as being expensive and difficult to do well (Hanley and Roberts, 2019; Bowen et al., 2012) and, as we highlighted in the discussion, this leads to small-scale and piecemeal analyses. Having a more standardised approach therefore enables more to be done for less. *Natural Capital Accounting* is, nowadays, becoming increasingly important to international and national governments, with growing use of the SEEA Ecosystem Accounting (SEEA EA) framework (UNCEEA, 2021).[22] Many of our estimates could be used to populate missing values from national or regional SEEA accounts. One can also 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. In the [much] longer term there may also be potential for a truly 'all agencies, all hazards' approach to Natural and other asset valuation, disaster risk reduction and community resilience.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

All data used in this article are provided in supplementary materials

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Appendix A. Supplementary data

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