The role of invasive alien species in shaping local livelihoods and human well-being: a review

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Abstract

Invasive alien species are a well-recognised driver of social-ecological change globally. Much research has focused on ecological impacts, but the role of invasive species for livelihoods and human well-being is less well known. Understanding the effects (benefits and costs) of invasive species on livelihoods and human well-being is important for guiding policy formulation and management. Here we review the literature on the role of invasive species in livelihoods to assess what is known, identify knowledge gaps and provide recommendations for future research. Literature was collected using key word searches and included both journal publications and grey literature. Findings suggest that most species provide a variety of benefits and costs for local livelihoods. Slightly less than half (48 %) of species studied had both substantial positive and negative impacts on local livelihoods (e.g. Australian *Acacia* spp. species ; *Camelus dromedaries; Lantana camara*; *Prosopis* spp.), with 37 % inducing mainly costs (*Chromolaena odorata; Lissachatina fulica; Opuntia stricta*) and 16 % producing mainly benefits (*Opuntia ficus-indica*; *Acacia* spp). Some species, such as *Acacia dealbata,* fell into different categories depending on the social-ecological context. Key benefits or services included the provision of fuelwood, fodder, timber and food products for local households and to a lesser extent supporting and regulating services such as soil improvement and shade. A number of species also provided cultural services such as recreation and spiritual values and provided many households with an opportunity to earn a cash income. However, invasive species also harm livelihoods and increase vulnerability through encroaching land and reducing mobility or access. They can also decrease the supply of natural resources used by households and reduce agricultural production which can result in losses of income and increased vulnerability. Furthermore, some invasive species were seen to have negative implications for human health and safety and reduce the cultural value of landscapes. Economic impacts on livelihoods as a result of invasive species were highly variable and very dependent on the social-ecological contexts. These negative implications can reduce resilience and adaptive capacity of households and communities thus increasing their vulnerability to change. Drawing on case studies we highlight that efforts for managing invasive species need to safeguard livelihood benefits while mitigating negative impacts. In concluding we highlight future research and policy needs on the topic of invasive species, livelihoods and human well-being.

Key Words

Biological invasions, ecosystem services; impacts; management; policy; social-ecological systems

1. Introduction

The movement of species out of their native range into new areas by humans has resulted in the rise of biological invasions (Mack, 2003). Introduced (exotic or non-native) species have to overcome a number of barriers to establish, naturalise, produce localised self-sustaining populations, and eventually spread naturally before they are considered as invasive (Pyšek et al., 2004). These invasive alien species are now recognised as one of the key drivers of human induced global environmental change as they affect biodiversity, ecosystem services and human well-being (Pejchar and Mooney, 2009). This makes research on invasive species important to guide policy formulation and management. However, prior research has predominantly been approached from a biological perspective, with some on economics of invasions, and a limited, but growing, understanding of the effects of invasive species on humans and their livelihoods and broader society (Vaz et al., 2017a).

Here we define livelihoods as the means of make a living (DFiD, 1999). Encompassed in the sustainable livelihoods framework are five domains all of which can be affected positively or negatively by invasive alien species (Chambers and Conway, 1992; Scoones, 2009). The first domain is the vulnerability context, which includes factors such as the presence of shocks and stressors and the social-ecological context (Adger, 2006). In this case invasive alien species can act as a shock or stressor and can alter trends within different social-ecological systems, many of which are known to induce long lasting regime shifts (Gaertner et al., 2014). They can also make individuals and communities more vulnerable to other shocks and stressors such as climate change, for example when invasive alien tress use up water increasing the severity of effects as a result of climate change induced drought or increase peoples exposer to natural hazards (Palmer et al., 2014). The second domain includes livelihood assets - namely, natural capital (the natural resource base, biodiversity and ecosystem services), social capital (social networks and organisations), human capital (knowledge and labour), physical capital (infrastructure, tools) and financial capital (money and credit). Here, invasive alien species can act as a resource improving natural capital and financial capital, or they can erode natural capital, negatively affecting financial capital and/or human and social capital. For example, many invasive species provide novel food sources which can act as a novel natural capital and can even provide incomes through sale of these food stuffs (Shackleton et al., 2011), but other species can invade fields thus reading crop output, and increasing labour times thus reducing incomes (Aslan et al., 2009; Engeman et al., 2010; Rajal and Cochard, 2016). The third domain encompasses transformative structures and processes - linking to governance, policy and institutions. Here factors relating to the governance of invasive alien species and land use, such as legislation, comes into play and can affect livelihood strategies and outcomes. For example, controlling useful species might negatively affect livelihoods (Middleton, 2012) or alternatively poor legislation and enfacement of control could lead to increased invasion thus increasing venerability and negatively affecting livelihoods. The fourth domain covers livelihood strategies - how one makes a living - in which invasive species can benefit or negatively affect livelihoods or alter their outcomes (Marshall et al., 2011; Rodgers et al., 2017). Lastly, the fifth domain represents livelihood outcomes – which refer to changes in vulnerability, capital, well-being, food security, governance and the like. Invasive alien species can affect livelihoods outcomes through reducing health and safety, increasing peoples exposure to hazards and shocks, improving or reducing food security and enhancing or degrading ecosystem services and thus can benefit livelihood outcomes or make livelihoods more vulnerable (Shackleton et al., 2007; Palmer et al., 2014; Rodgers et al., 2017). Therefore, invasive species can change livelihood vulnerability through being the catalyst inducing transformations at a number of levels as they can alter livelihood strategies and assets as well as change transformative structures and processes, thereby negatively or positively impacting livelihood outcomes and overall human well-being. This links closely to the provision of novel or better ecosystem services (benefits) or inducing novel ecosystem disservices (costs) (Shackleton et al., 2016a; Vaz et al., 2017).

The introduction of invasive alien species leads to alterations in the nature and quantity of ecosystem services or disservices supplied, which may then affect human well-being (Shackleton et al., 2007; Vaz et al., 2017; Potgieter et al. this issue). Components of human well-being include security, access to basic material to sustain a good life, heath, social relations and freedom of choice, which are factors mirrored in the sustainable livelihoods framework (MEA, 2005; Hannies-Young and Potschin, 2010; Smith et al., 2013). For example, invasive species can negatively affect human health through increasing the prevalence of disease or intensity of natural disasters like fires, can limit peoples' choices for income generation and can lead to the loss or insecurity in the supply of natural resources or capital important for sustaining a living and livelihood outcomes. In some cases invasive species can provide new resources that might improve the well-being of some primarily through providing novel livelihood outcomes (Shackleton et al., 2007; Mwangi and Swallow, 2008; Plamer et al., 2014; Rodgers et al., 2017).

The benefits and costs resulting from invasive species are determined by a multitude of social and ecological factors and can be very context specific (Shackleton et al., 2007; García-Llorente et al., 2008; Kull et al., 2011; Potgieter et al., this issue, Shackleton et al., this issue; Wald et al. this issue). A number of invasive alien species also have both benefits and costs which can lead to conflicts of interest between different stakeholders (Low, 2012; Zengenya et al., 2017; Villatoro et al. this issue). Therefore, management of these species needs to be carefully considered, seeking solutions that address the needs of both winners and losers. All these different combinations require that policy and management do not treat all invasive alien species in the same manner, but instead differentiate the types of invasive species according to their costs and benefits and according to the various stakeholders who experience these effects (Shackleton et al., 2007; de la Fontaine, 2013, van Wilgen and Richardson, 2014). Aside from the ecosystem services and disservices supplied by invasive species (Vaz et al., 2017; Potgieter et al., 2017), a number of factors also influence how invasive alien species will affect livelihoods, including the initial vulnerability of the community, the type and quantity of livelihood assets, invasive species traits, availability of resources, land tenure and the ecological and political context and other factors (Shackleton et al., 2007; Kull et al., 2011). This can make policy implementation and management complex; therefore it is crucial to understand the diverse role of invasive species for local livelihoods and society, to guide useful policy formulation and management options and implementation in the future, as well as to facilitate improved research going forward.

In this paper we review the literature to assess the role of invasive species in livelihoods and for human well-being. The purpose is to provide a synthesis of what is known and provide a platform to make recommendations for future work. In total 51 relevant case study sources were reviewed. Some of these sources had in-depth analyses of more than one invasive species, and therefore we had detailed case studies for 66 invasive species. We report on five key areas: (1) the invasive species, location, and social-ecological context of each case study (2) the methodology used for data collection, (3) the role of the invasive species in local livelihoods and their effects (benefits, costs and trade-offs), (4) local management operations and perceptions, and lastly, (5) policy and management recommendations considered by the authors of the paper.

2. Review methodology

We reviewed the literature to assess the role of invasive species for livelihoods and human well-being. We searched both academic and grey literature. The search words “invasive”, “alien”, “exotic”, and “livelihood”, “human well-being” and “well-being” “impacts”, “social” “benefits”, “costs”, “negative impacts”, “community”, “household”, “landowner”, “socio-economic and socioeconomic”, “soicio-ecologcial and socioecological”, “rural development” were entered into the ISI Web of Science and Google Scholar. We also used our personal databases, which included papers that were under review and student theses that are often not found via search engines. All the sources were then sorted for relevance and only case studies that directly assessed the role of invasive species in livelihoods were used. We also systematically went through the reference lists of all the included case study papers to search for additional sources. Opinion and review/synthesis papers were not included (e.g. Duncan et al., 2004; Pejchar and Mooney, 2009), and papers that did purely economic cost or cost-benefit analysis were also not included (e.g. Julia et al., 2007; Wise et al., 2012). Furthermore, a number of studies that listed potential benefits of all invasive species within a particular landscape, but did not delve into their effects on livelihoods, were not included (e.g. Semenya et al., 2012). We acknowledge that this review might have selection bias towards studies attuned to the invasion biology literature, which in general might focus more on negative impacts (Tassin and Kull, 2015). We also acknowledge that the term livelihood is more often used in reference to developing nations rather that developed ones and so might have a bias towards certain regions – we included terms such as “well-being” and other to try to account for this. We also recognise that the term ‘invasive species’ is biased towards Europe and ex-British colonial areas with large gaps in research for example South America (Speziale et al., 2012). In total 51 sources were reviewed (see Appendix 1 for a list of the case study sources). Some of these had in-depth analyses of more than one invasive species, and therefore we had detailed case studies for 66 invasive species.

Each paper was thoroughly read and data were extracted for a number of factors and variables, including: 1) Methods used for the study, by categorising the various approaches used, e.g. household questionnaire surveys, transect walks, key informant interviews, open forum workshops and others; 2) The location and social-ecological characteristics of the case study area such as the country, the land tenure, primary livelihood activities in the region, level of development in the region and others; 3) Background on the invasive species/s in the case study, which included the species name, functional group, date of introduction and the level of invasion; 4) The role of the invasive species in local livelihoods (benefits and costs). This included listing the benefits and costs mentioned in the paper. We then placed these into ecosystem services and disservice categories and captured the number of benefits and costs listed for each species in each paper. We also categorised each species on a scale of 1-5 based on its overall local community effects. For benefits, a score of 1 would mean that no or very few individuals were reliant or gained little benefit from the species, a score of 3 meant that a few individuals were very highly reliant or gained a lot of benefit from the species, or a large number of people were moderately reliant or gained a small benefit from the species, and a score of 5 would mean that most of the community gained substantial benefit from the species and was often highly reliant on it. The negative impacts were scored on the same scale; therefore a score of 1 meant very few individuals were impacted to a small degree, whereas a score of 5 meant that most of the community was sustainably negatively impacted by the species. We produced a scatter plot of these values (we did move scores by 0.1 or 0.2 as there was a large number of species that the same point, to show all species) to visually represent them within a matrix of costs and benefits based on the scoring above. This is a similar approach used to Zengenya et al. (2017). We also highlighted points of the same species or genera that fell in different quartiles. Based on the benefits and costs listed we then categorised each species into the four categorise outlined in Shackleton et al. (2007). This framework considers the trade-off between benefits and costs of invasive species and livelihood vulnerability over time. These four categories of invasive species include species with (1) no benefits and costs (undesirable, weakly competitive), (2) species with high costs and few benefits (undesirable, strongly competitive), (3) species with high benefits and low costs (desirable, weakly competitive), and (4) species with both high costs and high benefits (desirable, and strongly competitive). It must be noted that these species were categorised based only on the case study, and some might have been biased towards reporting only benefits or costs; 5) Information on local management operations. This included if the case study reported if locals were managing the invasive species in any way. If they were, we noted the type of management being implemented (e.g. mechanical control, chemical control, cultural control or utilisation); 6) Recommendations for policy and management made by the authors. For example, the push for management approaches such as biological control or utilisation, or highlighting that no management was needed; and 7) we captured any interesting findings that highlighted any factors that disposed communities to benefit or be negatively affected by invasive species and where there were contrasts between the same species in different social-ecological settings. This was also often done post the analysis of the results. This are discussed along with the main findings within the result section.

3. Results

Social-ecological context of the case studies

Studies on the role of invasive species in livelihoods emanate mostly from the developing world and communal rangeland areas, in particular countries in southern and eastern Africa and south-ease Asia. Even in Australia, a rich nation, a number of studies focused on rural communal aboriginal rangelands (e.g. Robinson et al., 2005; Vaarzon-Morel, 2008; Ens et al., 2016) with the same in Canada where the study focused on a first nations communal area (Bhattacharyya and Larson, 2014). In contrast, studies in the USA focused on private lands (e.g. Aslan et al., 2009; Palmer et al., 2014; Poudval et al., 2017). In many rich countries, studies more generally include strict economic cost models for estimating impacts on the economy, which is different from studies focusing on livelihood effects which are more holistic and often focus on vulnerability. Case studies came from 24 different countries (Figure 1). South Africa contributed the most case studies (18 %), followed by Australia, Ethiopia, India, Kenya and Madagascar and the USA accounting for 8 % respectively (Figure 1). The majority of case studies are in savanna and woodland ecosystems (54 %), and a few in forest, desert, Mediterranean and grassland biomes. Five case studies focused on freshwater ecosystems.

The majority of studies were conducted within community-based land tenure settings (63 %) with others taking place on multiple land tenures including both communal and private land, communal and protected areas, and just private lands. Almost all studies focused on communities that are reliant on subsistence agriculture and/or pastoralism, natural resource use, remittances, government grants, and unskilled labour although a small minority focused on private properties and commercial farms, particularly case studies in the USA (e.g. Palmer et al., 2014; Poudyal et al., 2017). Most studies (82 %) targeted a randomised subset of local households for data collection using questionnaires. A few employed techniques such as stakeholder mapping, key informant interviews, workshops and transect walks (e.g. see Bhattacharyya and Larson, 2014). Only 18 % of studies actively targeted specific stakeholder groups within the study community – this included those specifically involved in particular agricultural practices, fishing and the harvest of non-timber forest products (NTFPs) from particular invasive species (e.g see Ellinder et al., 2010; Shackleton et al., 2011; Pienkowski et al., 2015).

Figure 1: Global distribution of case studies . Circles increase in size and transparency based on the number of case studies in that country.

Invasive species investigated

Thirty-seven different invasive alien species were investigated, but the studies were biased towards taxa in a few genera (Figure 2). The majority of case studies focused on plants (80 %). This included 50 % focussing on trees and shrubs, 14 % on succulents, 5 % on vines and on aquatic plants and less than 5 % on grasses and herbaceous plants. *Prosopis* species were the most commonly researched taxa making up 22 % of the case studies, followed by *Opuntia* species (16 %), Australian *Acacia* species. (12 %), with *Chromolaena odorata, Equuas caballus, Lantana camara* and *Sus scrofa* (between 5-10 %)(Figure 2). A number of studies (12 %) focused on land mammals. Three studies included insects. Freshwater invertebrates and vertebrates, and land invertebrates were all only covered in one study each. This illustrates high bias towards research on particular functional groups, with many groups of invasive species not covered in livelihoods research.

The invasive species investigated were introduced as early as the 1700s up to as late as 2008. The majority were introduced between the 1950s and 1980s. Pathways for introduction differed substantially across the different case studies. Key pathways included introductions for afforestation/rehabilitation/agroforestry (30 %) often driven by international and local development corporation agencies or local government institutions trying to improve local livelihoods by combating land degradation, as in the case of *Prosopis* in many areas with introductions being promoted by the Food and Agriculture Organisation (FAO) and national governments (Mwangi and Swallow, 2008; Rogers et al., 2017). These species often have substantial conflicts of interest surrounding them and they provide both substantial benefits and costs. Many invasive species were also introduced by ornamental plant traders and colonists, often as ornamental species (22 %), a key one being *L. camara* (Kannan et al., 2014). To a lesser extent some invasive species were introduced initially for hedging, agriculture, aquaculture and commercial forestry. There were also some unique reasons for introduction, such as *Camelus dromedarius* for transport and *Cyprinus carpio* for recreation (Ellender et al., 2010; Vaarzon-Morel, 2008). About quarter (22 %) of species introductions were accidental, such as *CHromolaena odorata* in a number of areas (Siges et al., 2005; Rai et al., 2012) – and these species more often than not have less benefits than those introduced purposefully.

Case studies covered species that are invasive at a range of spatial scales and densities. The most commonly investigated species were generally present at high densities at regional and national levels, although a few also examined species that are relatively sparse or emerging (Sheil and Padmanaba, 2011; Luizza et al., 2016). In 79 % of case studies the invasive species was identified as increasing, either as a direct report from local respondents, through citing other work or through observations or mapping by the authors. A small number (6 %) of case studies identified that the populations of invasive species were either decreasing or fairly stable (e.g. Shackleton et al., 2007). The remainder did not give an indication on the temporal dynamics of the invasive species.

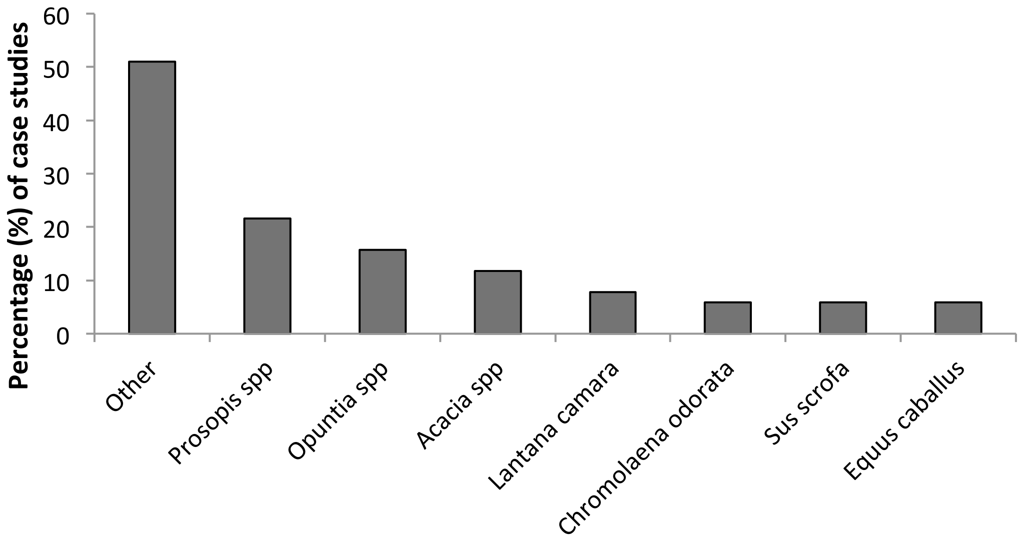

Figure 2: Prevalence of case studies on different taxa. "Other" refers to all the genera that were only covered in one or two studies.

Livelihood impacts of invasion (benefits and costs)

A large number of case studies listed benefits of invasive species for local livelihoods (79 %), but most case studies (86 %) also listed livelihood costs. For 14 % of case studies, the invasive species investigated either did not have livelihoods costs, or they were not mentioned in the study (e.g. *O. ficus-indica* in Shackleton et al. (2011)). This highlights that most studies identified both costs and benefits, even though many of the studies were probably biased to reporting costs which is common in the ecological domain of invasion science (Tassin and Kull, 2015). A number of studies, such as Ellender et al. (2010), Shackleton et al. (2011), Rai et al. (2012) and Pienkowski et al. (2015), only investigated benefits even though the species may have had costs for livelihoods as well. The mean number of benefits reported per invasive species case study was three as opposed to five for costs.

These benefits and costs have a number of influences on sustainable livelihoods (Chambers and Conway, 1992; Scoones, 2009). Many invasive species negatively impact livelihoods assets as they require additional labour and financial resources to manage them thus depleting human, physical and financial capital. A number also increase exposure to natural hazards and shocks. This can negatively impact livelihood strategies and outcomes though reducing incomes, food security, adaptive capacity and reducing well-being thus increasing induvial or community vulnerability contest. Alternatively, some invasive alien species can provide novel resources in the form of additional natural capital, which can change livelihood strategies, i.e. provide fishing as a new option, and thus improve livelihood outcomes, through creating employment or income generation and which can reduce the overall vulnerability of an invidual or a community and increase adaptive capacity. A number of species provide both benefits and costs in different contexts which make it important to understand trade-offs for decision making.

Livelihood benefits

A large range of benefits were identified across the different case studies, linked to different provisioning, regulating and cultural ecosystem services. Many invasive alien species provide natural resources or NTFPs which benefited livelihoods as they provide improved or novel livelihood assets such as natural capital and the ability to create jobs to improve financial capital (Chambers and Conway, 1992; Scoones, 2009). This commonly included the provision of fuelwood, fodder, food products, timber and medicinal products (Figure 3). Invasive species also provided other benefits for livelihoods such as soil improvement though green manure and nitrogen fixation, live fencing, and cultural services, such as recreation and aesthetic values (Figure 3). Economic benefits through providing resources for income generation was also highly important for livelihoods (Figure 3).

The proportion of people using invasive alien species and the importance of the benefits for livelihoods varied substantially between the case studies and different invasive species. For many species (44 %) the proportion of the community benefiting and the importance of the benefit was ranked as zero or fairly low (e.g. *Alternanthera philoxerodies* (Keller et al. 2017), *Anoplophora glabripennis* (Palmer et al. 2014), *Cryptostegia grandiflora* (Luizza et al. 2016), *M. pigra* (Rijal and Cochard, 2016) and others). For a number of species (24 %), the importance of the benefits for livelihoods was moderate – meaning a small number of community members benefited greatly or a moderate number benefited to a small extent (E.g. *Piper aduncum* (Siges et al., 2005), *Cypricnus carpio* (Ellinder et al. 2010). Just under one-third of species (32 %) were reported to be highly important for local livelihoods, meaning that a large proportion of the community benefited and the benefits were important to sustain local livelihoods and human well-being. For example, at a site in in Malawi, all households used *Prosopis* for fuelwood, which was their key source of energy cooking and heating, and 44 % of households made some income through selling *Prosopis* fuelwood (Chikuni et al., 2004). Similar levels of *Prosopis* use are mirrored in studies in other regions such as Kenya, Pakistan and South Africa (Mwangi and Swallow, 2008; Kazmi et al., 2009; Shackleton et al., 2015). Similarly, Australian *Acacia* species in South Africa and Madagascar were used as a fuelwood by all residents in the case study sites, and 19 % relied on it as an income source in South Africa (de Neergaard et al., 2005; Kull et al., 2007). *Acacia dealbata* is a particularly important resource for communities in the Eastern Cape of South Africa as the availability of other trees in the high altitude grasslands is low (Ngorima and Shackleton, this issue). *Opuntia ficus-indica* and *Prosopis* were used as a fodder and food source by most members of the community in a number of countries where they are invasive (Beinart and Wotshela, 2003; Larsson, 2004; Shackleton et al., 2011, 2015; Mwangi and Swallow, 2008). The benefits from *Cenchrus ciliaris,* a grass useful for grazing, has led to actively planting and facilitation of its growth by many farmers in Australia (Marshall et al., 2011).

The majority of studies listed benefits with most also quantifying or giving an indication of the number of users, however, only a few reported financial values of all or some benefits. Furthermore, the financial estimates of benefit were derived in a number of different ways that are not comparable. From reports across a number of countries and invasive species, annual direct monetary benefit per household from invasive species was fairly low. However, for many studies the benefits from invasive species are reported from rural and often underdeveloped areas and consequently the benefits from invasive species may represent a substantial contribution to incomes and livelihoods for some households or groups. For example, Shackleton et al. (2011) describe households selling *O. ficus-indica* fruit in South Africa earning between US$ 150 and 300 per year, typically representing 9 - 30 % of annual cash income for the household. However, less than 1 % of households engaged in selling the fruit. In contrast, the sale of *L. camara* based products in certain villages in India accounted for 46 % of annual household income for sellers, and with more than half the households participating in such sales (Kannan et al., 2014).

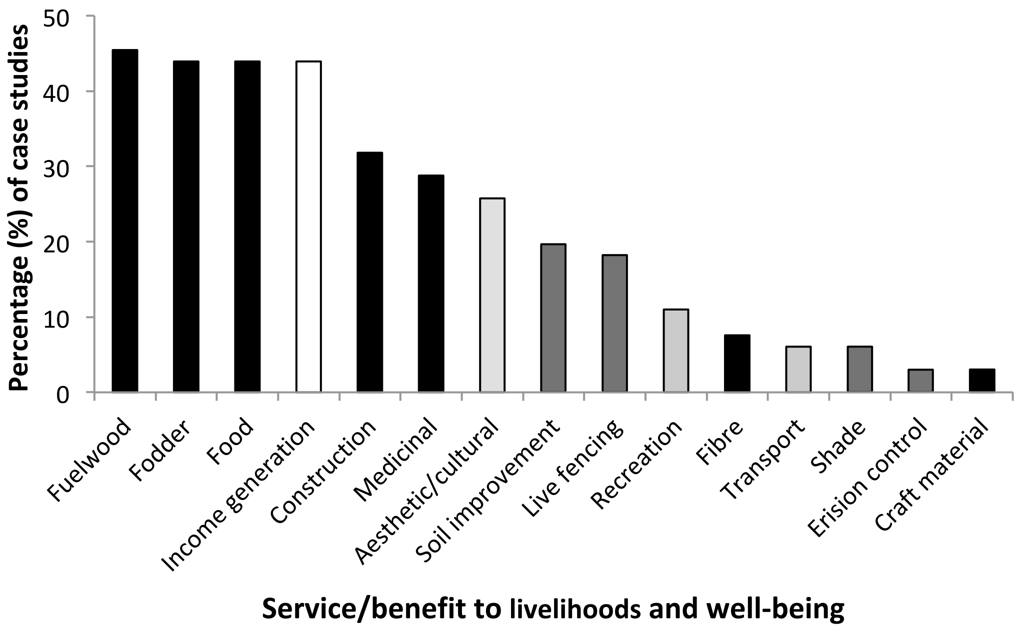

Figure 3: Prevalence of the different benefits to livelihoods derived from invasive species. These benefits are categorised by ecosystem services and elements of human well-being. Black bars represent provisioning services, dark grey bars supporting and regulating services, light grey bars represent cultural services and the white bar is an aspect of human well-being that is derived from ecosystem services provided by invasive species.

Livelihood costs

A range of negative impacts, costs, or ecosystem disservices may be incurred as a result of invasive species, resulting in reduced human well-being and increased livelihood vulnerability often though the loss of livelihood assets and livelihood outcomes (Chambers and Conway, 1992; Scoones, 2009). The most commonly reported negative impacts included the loss of NTFP supply (generally linked to the loss of biodiversity and altered plant community structure – loss of natural capital), reductions in crop yields (directly or due to less land available to cultivate), loss of grazing and livestock production, impacts on human mobility and access to land, health and safety issues, loss of incomes (financial capital) and increased labour times (human capital) (Figure 4). A number of plants also have negative cultural effects (see Back et al. this issue). Unlike the benefits, the costs incurred to livelihoods as a result of invasive species were much more wide-reaching. Only 20 % of invasive species had little impact for communities or impact were not mentioned (e.g. *Cecropia peltata* (Sheil and Padmanaba, 2011), *C. cilliaris* (Marshall et al., 2011), *Grevillea banksii* (Kull et al., this issue), *O. ficus-indica* (Shackleton et al., 2011) and others). A minority, 12 % had moderate impacts (e.g. *Bubalus bubalis* Robinson et al., 2005; Ens, 2016), *Tithonia diversifolia* (Witt et al., in press) and others) and the majority (67 %) had far reaching negative impacts for most of the community (e.g. *A. glabripennis* (Palmer et al., 2014), *Makania micrantha* (Rai and Scarborough, 2015), *O. stricta* (Shackleton et al., 2017b), *Prosopis* (Chikuni et al., 2004; Laxen, 2007, Maundu et al., 2009), *S. scrofa* (Poudyal et al., 2017) and others). The majority of case studies (85 %) did not quantify costs monetarily, but just listed them along with some qualitative measure or ranking of how communities were impacted. For studies that did quantify costs monetarily, it was done using a number of approaches, including costs to control the species, or direct monetary loss such as livestock death or losses in agricultural production (Engeman et al., 2010; Rijal and Cochard, 2016; Ngorima and Shackleton, this issue; Poudyal et al., 2017).

Financial costs varied widely, depending on the invasive species, the method used to quantify monetary impact, economy of the country (cost of labour) and land tenure. The mean value of the financial costs estimated across the different case studies was highly variable, and dependent on the methodology used to estimate the costs and the social-ecological context. For example, commercial livestock farmers in South Africa paid substantially more, US$ 2 000 per annum, to manage *Prosopis* (Shackleton et al., 2015) as compared to rice farmers in Cambodia at US$ 20 per annum (Rajil and Cochard, 2016). This said, the South African farmers were much better off financially as they operate much larger commercial farms compared to their Cambodian counterparts, making comparisons difficult. Comparison would therefore be easier if costs were expressed as a proportion of annual cash and non-cash income. Costs also often varied between households with different livelihood strategies, for example many invasive species impact heavily on livestock production, therefore communities and households whose primary livelihood strategy is subsistence pastoralism would be impacted more than other people (Shackleton et al., 2017ab). This is similar for aquatic weeds where fishermen are more heavily impacted - although other community members are impacted indirectly though the presence of these water weeds (Keller et al., 2017). Furthermore rural households seem to carry a heavier impact of invasive species than urban households, as they rely more on natural resources and provisioning ecosystem services for livelihoods (Shackleton et al., 2015). In some cases, however, the impacts of invasive species can also be very high in more urban areas as with the case of *A. glabripennis* destroying trees in urban areas of Worcester, Massachusetts (Palmer et al., 2014). Differential impacts may also be evident between genders, especially with issues around health and safety. For example, rape of women within dense *Acacia* invasions in South Africa has been highlighted as a major issue (de Neergaard et al., 2005; Norgaard, 2007; Ngorima and Shackleton, this issue). Density of invasion plays a key part in determining who and the proportion of households negatively impacted (Shackleton et al., 2007; Shackleton et al., 2016).

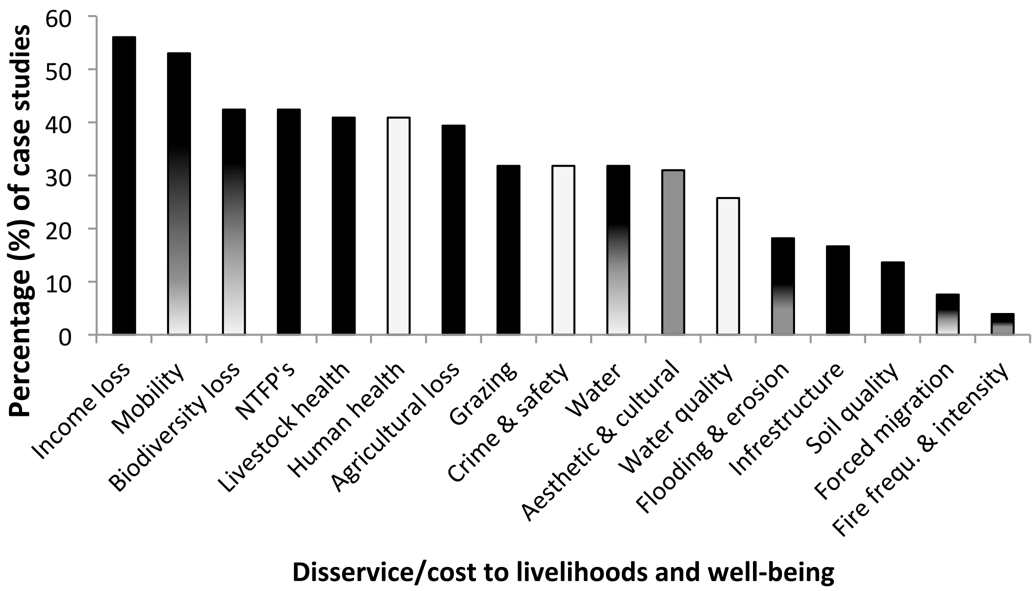

Figure 4. Prevalence of the costs to livelihoods as a result of invasive species. These impacts are categorised using the Shackleton et al. (2016) ecosystem disservices categories. Black bars are economic disservices, light grey are health and safety disservices, and dark grey bars are cultural disservices and multi-coloured bars are cross cutting impacts that fall into multiple categories with rough estimates on levels of impacts according to the different disservice categories.

Benefits vs costs

Using the Shackleton et al. (2007) framework for assessing the impacts of invasive species on livelihoods, all the species were divided into one of four categories based on their impacts (linked to their competitive ability and potential to increase livelihood vulnerability) and benefits (if any) that they supply to livelihoods (see Figure 5). Approximately one-fifth (16 %) were categorised as "desirable and weakly competitive" – although this number might be larger as there is almost certainly a selection bias towards studying species with large impact (Figure 5). A species commonly categorised as "desirable and weakly competitive" was *O. ficus-indica*, as it was seen to occur at stable densities and provide a number of benefits, particularly food and fodder (Larsson, 2004; Shackleton et al. 2011). These invasive species generally have a positive influence progress toward sustainable livelihoods though improving livelihood assets, providing new livelihood strategies and outcomes, improving adaptive capacity thus reducing the overall level of livelihood vulnerability (Chambers and Conway, 1992; Scoones, 2009). In the past this species may have fallen in the "desirable, strongly competitive" category but it in many regions it is now under biological control and densities have decreased and stabilised (Beinart and Wotshela, 2003). In Madagascar *Acacia* species also fell into the "desirable and weakly competitive" category as locals viewed them as having high benefits, low spread rates and minor impacts (Kull et al., 2007), whereas, in South Africa they fell into the "desirable and strongly competitive" category (de Neergard et al., 2005; Shackleton et al., 2007; Ngorima and Shackleton, this issue) (Figure 5). This may be because land use intensity is higher in Madagascar and so it is difficult for Acacias to develop large stands, whereas it is possible for them to do this in rangeland areas of South Africa. Similar differences were seen in the case study for *L. camara* in India (Kannan et al., 2014), showing more benefits than the one in Uganda (Shackleton et al., 2017a), where in India an NGO has helped communities set up furniture making businesses using *Lantana* instead of native bamboos (Figure 5). This therefore suggests that the social-ecological context of the area plays an important part in determining benefits and costs of invasive species.

Many of the invasive alien species (48 %) fell into the “desirable, strongly competitive” category, again which may be as a result of selection bias. These species resulted in both benefits and costs to livelihoods and human well-being. Species in this category included *C. dromedaries, L. camara, P. aduncum, Prosopis* and *Acacias* among others (de Neergaard et al., 2005; Shackleton et al., 2007; Mwangi and Swallow, 2008; Maundu et al., 2013; Shackleton et al., 2015; Ngorima and Shackleton, this issue). *Lantana camara* in southwest India has been described as a driver of social-ecological change through its increased use as a substitute for bamboo that had been over-harvested, so this species, unlike others, may have increased benefits to livelihoods over time in some areas (Kannan et al., 2014). Similarly, *E. caballus* benefits and costs vary between stakeholders and in different contexts (Robinson et al., 2005; Bhattacharyya and Larson, 2014; Ens, 2016). Species that fall in the category of both benefits and costs are often particularly complex when it comes to policy formulation and managing implementation as they can lead to conflicts of interest between different stakeholders (van Wilgen and Richardson, 2014; Zengenya et al., 2017).

Lastly, a number of species posed substantial costs without any or very few benefits, such as *Azolla cristata, Aceria guerreronis*, *Centaurea solstitialit*, *Cryptostegia grandifloris, Lissachatina fulica, Chromolaena odorata, Mikania micrantha, M. pigra,* and *Opuntia stricta* as well as a number of other species (Larsson, 2004; Aravindakshan, 2011; Alsan et al., 2009; Rai et al., 2012; Luizza et al., 2016; Rajal and Cochard, 2016; Keller et al., 2017; Shackleton et al., 2017b; Stronge, undated). These species should be prioritised for management, as they do not benefit human well-being and in many instances substantially increase livelihood vulnerability through impacting different forms of human capital and livelihoods outcomes as well as reduce adaptive capacity (Chambers and Conway, 1992; Scoones, 2009; Palmer et al., 2014).

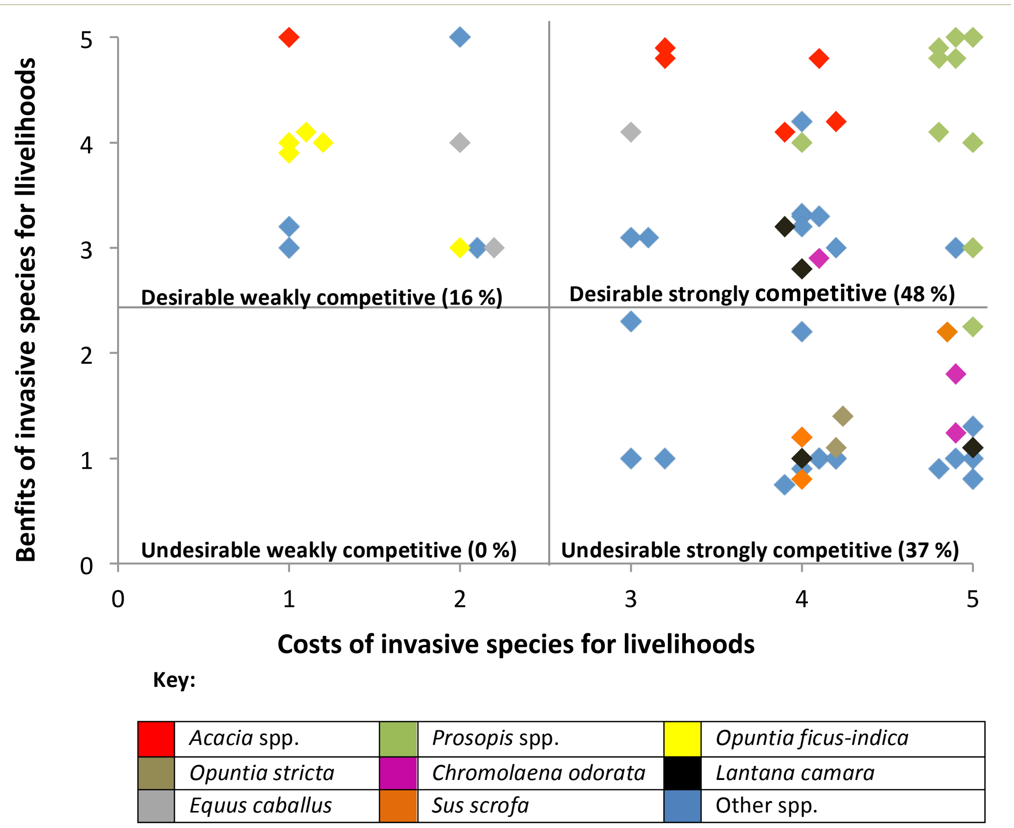

Figure 5: The trade-offs in the overall benefits and costs to local livelihoods on a scale of 1 to 5 (Similar to Zengeya et al. (2017). Different colour points represent the same species or genus in different case studies (see key). The quartiles are based on the four categories outlined in the Shackleton et al. (2007) framework, whereby species can be: Undesirable – Weakly Competitive (have low befits and costs); Undesirable – Strongly Competitive (have low benefits and high costs); Desirable – Weakly Competitive (have high benefits and low costs) and Desirable – Strongly Competitive (have high benefits and costs).

Management of invasive species to reduce vulnerability and improve well-being

The majority of case studies (76 %) mentioned some attempts at localised management by community members and land owners to mitigate the negative effects of the invasive species on their livelihoods which can drain human and financial capital, but at the same time reduce vulnerability from invasive species. Of those studies mentioning control by locals, the most common control approach used was mixed methods (51 %) including two or more methods simultaneously (mechanical (pulling, cutting, trapping, shooting etc.), chemical, biological control, utilisation and land use adaptation). A number also mentioned the use of only mechanical control methods (16 %), utilisation (24 %), and alterations or adaptions to land management practices (8 %) which included fencing, fire and altered grazing practices as forms of control. Chemical control was used for some invasive plants (18 %) along with other approaches, however, the expense of chemicals was likely to be beyond the means of poor, rural communities living in communal lands described in most of the case studies. Each of these methods have associated positive and negative aspects, for example there are contamination and toxicity concerns with the use of chemicals (Tassin, 2017), and reliance on use of the invasive species may potentially have the perverse consequence of promoting their spread (van Wilgen et al., 2011; Kannan et al., 2016). Most case studies (79 %) reported continued spread and increasing densities despite some attempts at management by the local community or government aided projects. Some studies did, however, mention that the invasive species population was stable or decreasing – in particular for *O. ficus-indica*. One study notes a decrease in abundance of the water weed *Eichhorina crassipes* for some time due to biological control, but a subsequent increase with time (Opande et al., 2004).

Opuntia ficus-indica represents an interesting case study of how effective management has essentially taken this invasive species from a "desirable, strongly competitive" species with overall negative impacts for livelihoods and shifted it towards being a "desirable, weakly competitive" species. It has become a desirable natural capital resource which improves livelihood outcomes and reduces vulnerability (Figure 5). In the case of South Africa, this was done though an effective biological control programme launched by the government in the 1930s (Beinart and Wotshela, 2011). At the height of invasion *O. ficus-indica* covered almost 1 million ha and had substantial impacts on livestock production and human mobility, reducing well-being and increasing livelihood vulnerability. Two biological control insects, *Dactylopius opuntiae* (cochineal) and *Cactoblastis cactorum* (moth), were released and within a few decades *O. ficus-indica* cover was reduced to under 100 000 ha allowing for a much larger supply of benefits from other services in the landscape and *O. ficus-indica* itself, due to increased accessibility and a change towards positive perceptions for the plant (Zimmerman and Moran, 1991). Some communities even mentioned that they would not mind if densities of *O. ficus-indica* increased slightly due to the benefits it provides (Shackleton et al., 2007). This successful biological control programme is widely recognised globally for reducing costs and improving benefits for livelihoods. However, in contrast, in southern Madagascar the biological control of *Opuntia monocantha* in the 1920s and management efforts on other *Opuntia* taxa in recent decades have had controversial effects on local communities, who had adapted their livelihoods to the use of the cactus (Binggeli, 2003; Middleton, 2012; Kaufmann, 2008). This suggests that social-ecological contexts play a crucial role in determining the role of invasive species for livelihoods and thus management implementation need to be considered thoroughly beforehand. A number of other successful biological control programs for *Acacia* spp. in South Africa may also reduce densities to an extent where the benefits might outweigh the costs for livelihoods in the , such as?? (van Wilgen et al., 2012).

In the majority of case studies (75 %), the authors discussed some form of management or policy response to reduce negative effects of the invasive species on livelihoods and possibly also improve benefit supply. For plants, a common recommendation was to research and introduce biological control agents (44 %). This approach is probably preferred by many researchers due to its usually cost-effective nature as well as its low, long-term maintenance needs (Page and Lacey, 2006). However, in some cases biological control agents may fail to establish. In Kenya, the majority of agro-pastoralists had noticed the biological control agent on *O. stricta*, however, only 36 % believed it was fully safe and was purely beneficial, with the remainder being unsure about biological control management (Shackleton et al., 2017b). However, all respondents were happy with biological control if it posed a safe and cost-effective way of reducing undesirable invasions. Biological control for *C. odorata,* a species negatively affecting livelihoods, has been successful in Indonesia and Papua New Guinea and has substantially benefited local livelihoods through improving crop production and returns from agroforestry (Zachariades et al., 2009; Day et al., 2013). For all biological control efforts in Australia – including a large number of agents that are not having a substantial effect at reducing the cover of invasive species or failing to establish - the overall benefit to cost ratio is still 23:1, and other countries should look into this as a means to control invasive species to improve local livelihoods and reduce vulnerability (Page and Lacey, 2006).

A number of studies (32 %) suggested that the state and local communities need to promote awareness and more participation in mechanical and chemical clearing. A third of cases (36 %) also advocated for increased use (utilization) as a control measure for invasive species with substantial benefits and included both plants and mammals (e.g. Robinson et al. 2004; Mungatana and Ahimbisibwe, 2012). This option is only limited to useful species (those providing natural capital) and remains controversial (van Wilgen et al., 2011). To a slightly lesser extent alterations or adaptions to land management practices were discussed as a way to reduce the livelihood impacts of invasive species (19 %). This may be because such methods are usually highly context specific and difficult to implement. For example, use of fire to manage *Prosopis* in South Africa is not possible as the invasive hybrids are fire resistant. However, such methods can also show success, such as fencing off of billabongs in Australia to prevent damage from invasive animal species (Ens et al., 2016) and mowing, burning, alterations to grazing and maintaining natural areas to reduce some plant invasions (Aslan et al., 2009, Marshall et al., 2011). The recommendations of most studies focused on the need for large-scale control programs driven at a national or regional level, while fewer stressed the importance of smaller-scale, localised management operations to reduce the impacts of invasive species for livelihoods. In a more unique discussions Palmer et al. (2014), call for building leadership and trust, improved co-management and to promote institution flexibility at time of uncertainty to improve adaptive capacity in responses to invasions to improve resilience and recue vulnerability. The ideas of co-management are also mirrored in Robinson et al. (2004).

4. Future directions /needs

Studies on the livelihood effects of invasive species are increasing and have been useful to understand the suite of different benefits and costs of invasions, in particular economic ones, as well as to provide evidence and approaches to aid management actions where necessary. In particular, this helps to understand how invasive species affect livelihood assets, livelihood outcomes and vulnerability contexts, and the role of management in transforming these relationships. Research remains skewed towards the developing world (as the sustainable livelihoods concept is normally applied in the context of developing countries) and future research could incorporate more novel methodologies. Here we highlight four crucial needs and areas for future research as well as discuss implications for policy, governance and management.

(1) Improve understanding over larger temporal, spatial and social-ecological scales

Based on this review (Figure 5) and other work such as Shackleton et al.'s (2007) framework, it is clear that the role of invasive species in local livelihoods is dynamic in space and time. More projects assessing the role of invasive species in livelihoods over time, as well as comparing between different stakeholders and contexts will provide a lot more insight. The majority of studies to date represent a single time point – and do not provide any indications of the roles the invasive species played in the past, or how changes in context may influence use and livelihood benefits or costs. Some exceptions exist, such as Shackleton et al. (2017a) that compared knowledge, perceptions and livelihood impacts between communities where *C. odorata* was a recent arrival and had a lower density (on the invasion front), compared to communities where the invasion was long-lived and at higher densities. This study revealed higher knowledge and impacts in the site with longer exposure to and experience of *C. odorata*, which accords with the Shackleton et al. (2007) model that suggests negative impacts and vulnerability increase and benefits decrease with time if populations grow unchecked. Similarly, though Udo et al.(this issue), highlight that gorse on Reunion went “From useful to invasive” over time with changes in effects to livelihoods and society. Converse trends also exist, for instance where through effective biological control *O. ficus-indica* populations have decreased, resulting in improved livelihoods for commercial farmers and rural communal villages in South Africa, with the net benefits of the species currently likely to be positive (Beinart and Wotshela, 2003). Case studies investigating the relationship between the duration of invasion with the passing of different thresholds of livelihood and human well-being impacts would provide key insights (Shackleton et al., 2017a). Another important area is to link species distribution modelling and future spread models and understanding future livelihood effects.

A number of species have been studied in different spatial and social-ecological contexts and our review has revealed that their role in livelihoods may vary considerably within different contexts (Figure 5). For example, Australian acacias did not have many negative impacts on livelihoods in Madagascar where land use intensity is high compared to the rural, communal lands of South Africa where land use intensity is a lot lower, resulting in much denser stands and higher impacts (Kull et al., 2007; Shackleton et al., 2007). Similarly, crime came out as a major issue relating to invasive Australian acacias in South Africa but much in other countries, and relates closely to prevailing social-political aspects in that country.

The results of the review clearly show that the study of invasive alien species’ effects on livelihoods has been focused on developing countries. Furthermore, in both developed and developing countries there is a strong focus on communal, rural rangelands, (e.g. Vaarzon-Morel, 2008; Bhattacharyya and Larson, 2014) – although there are exceptions (e.g. Steele et al., 2006; Palmer et al., 2014) (Figure 1). Relatively, there are fewer case studies from developed nations – where studies tend to focus on the economic costs of invasives to particular productive land uses, rather than on peoples’ livelihoods *per se*. A number of invasive species in Europe and North America are likely to affect local livelihoods. For example, *Heracleum mantegazzianum* and *Ambrosia* spp. have significant human health impacts, *Bromus tectorum* has negative implications for livestock production while *Robinia pseudoacacia* provides wood, honey and cultural services but also has negative implications for biodiversity and regulating ecosystem services, *Oncorhynchus mykiss* has economic benefits for livelihoods and has substantial recreational value along with many other examples of invasive alien species – therefore all influencing lives and livelihoods (DiTomaso, 2000; Duncan et al., 2004; Thiele and Otte, 2007; Montagnani et al., 2017; Vítková et al., 2017). There are therefore a large number of invasive species and contexts requiring greater depth of understanding of their role for people’s livelihoods and well-being. There needs to be a push for more research on this topic in the global north as there will be similar dynamics but possibly many differences that need to be teased out and understood.

Linking to this, most studies were conducted on rural lands under communal tenure, and consequently there is scope to better understand livelihood effects in tenure regimes such as private agricultural land and in-and-around urban areas. One way forward would be to focus on a globally widespread invasive species to allow an international, comparative assessment of its livelihood impacts and roles in different contexts and under varying tenures.

(2) *Greater distinction between different stakeholders (lives and livelihoods) and direct and indirect impacts*

To date, most studies have grouped all stakeholders together or only assess one particular group without any differentiation or comparison between different groups - with a few exceptions (e.g. Kannan et al., 2014; Shackleton et al., 2015). These studies show that the livelihoods of different groups are affected to varying degrees and in different ways. For example, the benefits and negative impacts of *Prosopis* invasion were more prominent for private famers as compared to urban city dwellers (Shackleton et al., 2015). Benefits also differed for these groups, for example cultural services were of more important in urban settings compared to farmlands. Similarly Kannan et al. (2014) split the study community into two groups (*L. camara* users and non-users). Their findings unsurprisingly illustrate that the overall costs of *Lantana* are higher for non-users, whereas users derive significant benefits, with *Lantana* being an important resource for the poorest in tose communities. Therefore, future research needs to differentiate between different stakeholder groups to ascertain who carries the costs and who gather the benefits, why and how this has changed with time.

Furthermore, the effects of invasive species might differ between demographic groups. This relates to subdividing stakeholders within the same population more finely (e.g. men and women, agro-pastoralists or pastoralists; wealthy and poor; those with land and those without). For example, issues relating to crime and personal safety around dense invasions, in particular rape, are much higher for women than for men (de Neergaard et al., 2005; Ngorima and Shackleton, this issue). Similarly, the provision of natural resources by some invasive species might be more important for poorer households who have fewer options for other sources of income provision, than better-off households (Kannan et al., 2014).

Linking to this is the need to consider what we call "direct" and "indirect" livelihood benefits and costs for different parties and demographic groups. For example, hypothetically, fishermen may experience increased vulnerability due to the negative effects of an invasive fish. However, this has indirect effects on food security and fish prices for other people. It also has potential indirect effects for fishermen's families, though issues like food security and the ability to send their children to school.

(3) *Better understanding of the social dimensions of livelihoods relating to culture, cultural services and human well-being*

Most studies focus on the economic impacts of invasive species for livelihoods and the value of provisioning services (Figure 3 and 4). Although these might be the primary benefits and costs for livelihoods induced by invasive species, these also might be the easiest metric to measure as argued by Pejchar and Mooney (2009). Knowledge and understanding of the effects of invasive species on cultural services and well-being is limited and often overlooked. In many cases cultural impacts and benefits seem to be added as an interesting anecdote that was stumbled upon during the research with not much further consideration or discussion (e.g. Shackleton et al., 2007). This is common throughout ecosystem services and environmental valuation research where understandings of cultural services are lacking (Chan et al., 2012). However, many invasive species have major influences on cultural services inducing substantial benefits or costs and consequently, novel methodologies are required to gain more insight (Vaz et al., 2018).

Many invasive species also alter cultural practices and livelihoods. For instance, *Lates niloticus* has changed culture and human well-being around Lake Victoria through the alteration of traditional fishing practices whereby local, small-scale fishers were replaced with commercial fisheries and later the collapse of the entire fishing industry led to a number of social ills such as substance abuse, prostitution and the spread of HIV/AIDS (Molony et al., 2007). In another case, *Opuntia* species in Madagascar permitted mobile pastoralists to adopt settled agro-pastoral practices, thus fundamentally changing livelihood and cultural practices (Kaufmann, 2008). This area of work needs more investigation. Some have reported the local adoption of invasive species as culturally or spiritually important, such as *O. ficus-indica* as “a plant of our ancestors” (Shackleton et al., 2007) or the worship of *L. camara* bush by a community in India (Kannan, 2011).

Similarly, there are cultural differences in the way communities view invasive alien species and how they should be thought about, classified, and managed. These differences should be addressed and given voice (see Norgaard, 2007; Bhattacharyya and Larson, 2014; Ens et al., 2016; Bach et al., this issue). Such sentiments will clearly influence any suggestions or actions to control the species in such landscapes.

(4) More integrative methodologies

The majority of studies reviewed used randomised or semi-randomised questionnaire surveys to elicit information on the role of invasive species on peoples’ livelihoods and well-being. This is the most common methodology used in studies that investigate stakeholder actions, knowledge or perceptions in the field of invasion science (Shackleton et al., this issue). Therefore, currently the norm is to present quantitative data collected via these surveys. This is useful and should not be limited. However, there are a number of other methodological frameworks and approaches that could enrich the insights obtained and result in improved and more holistic understandings on the topic. More participatory approaches that allow for flexibility in the methods can often provide interesting and unforeseen findings. They can also improve co-design of research questions and improve social learning (Shackleton et al., this issue). These approaches might also better elicit different social and cultural impacts and viewpoints - a need suggested above. For example Luizza et al. (2016) successfully used community meetings and participatory mapping to understand the role of the vine *C. grandiflora* in peoples’ livelihoods in Ethiopia, which also produced maps for targeting emerging populations. Ngorima and Shackleton (this issue) used participatory ranking, mapping, transect walks and focus-group discussions to elicit greater depth of understandings and personal stories regarding the role of *A. dealbata* in local livelihoods – this is also mirrored in other studies (Bhattacharyya and Larson, 2014; Luizza et al., 2016).

There is also lot more scope for more narrative case studies using in-depth stories from key informants to better understand the role of invasive species in livelihoods and well-being, seeking insights into causes, processes and effects beyond just survey results. Other methodological options include discourse analysis with regards to in-depth interview or written data (Cottet et al., 2015). To transform the largely ecological visage of invasion science requires that research is more interdisciplinary and based on a range of methodologies and disciplinary framings.

(5) Policy, governance and management implications

Based on the findings there are a number of considerations that should be made in the future relating to policy, governance and management of invasive species to ensure sustainable livelihood strategies and outcomes, improve adaptive capacity and to ensure that that communities are not made more vulnerable by invasive alien species.

First, we highlight the need for improved and more rigorous risk assessments in the future (Leung et al., 2002; Keller et al., 2007), for introduction of species aimed to improve local livelihoods. A number of species like *Prosopis* were purposely introduced by governments and agencies such the Food and Agriculture Organisation (FAO) to aid development and rural livelihoods in many counties during the mid-19th century prior to a real understating of biological invasions. This has led to major negative implications for many rural communities and local livelihoods and well-being and has markedly increased local vulnerability contexts in a number of countries around the world (Shackleton et al., 2014). Despite the growing understanding of the mixed implications of introducing species that might become invasive, many rural development agencies continue to introduce species without risk assessments. For example *Prosopis* is still being actively planted in Myanmar by government institutions (Aung and Koike, 2015). Similarly, a number of known invasive species such as *Jatropha spp.* are being planted and promoted for biofuel in Africa (Witt, 2010). Species that are deemed high risk in particular contexts should not be introduced and alternatives should be sought.

Second, we highlight the need to weigh up the costs and benefits for livelihoods of specific invasive species in specific places prior to their management. In the past not doing so has led to negative implications for local livelihoods, as seen with *Oputia* in Madagascar (Binggeli, 2003; Middleton, 2012; Kaufmann, 2008). The importance of this is increasingly recognised, as seen in the work that was commissioned relating to understanding the livelihood effects of *A. dealbata* prior to the implementation of biological control program (Ngorima and Shackleton, this issue). That study showed that local communities do want a lower abundance of the invasive tree, but not total removal because it is widely used as a resource. These sorts of studies can help guide the choice of control agents and approaches. In cases where species have large benefits with very little costs (Zengenya et al., 2017; also see Figure 5), policy could consider accepting these species as part of novel ecosystems, after rigorous costs benefit analysis. Assessing the livelihood effects and trade-offs also aid with planning and prioritisation of control – especially to ensure effective use of limited resources (Shrestha et al., this issue)

Thirdly, in the numerous cases where invasive species have negative impacts on livelihoods and human well-being, we would advocate the promotion of biological control as a cost effective management approach. This is particularly the case where invasions have a large extent. Properly researched biological control can be a cost effective and sustainable control approach (van Wilgen et al., 2012). For example, biological control of *O. ficus-indica* invasions has led to it providing net benefits for livelihoods in South Africa. The successful biological control of *C. odorata* has greatly improved agroforestry production for substance farmers in Indonesia and Papua New Guinea (Day et al., 2013), thus improving livelihood outcomes and reducing local vulnerability contexts. In the case of invasions with limited geographic extent, a community-based management or eradication approach in support of livelihoods is likely to be a preferred option in many contexts.

Fourthly, there needs to be a push for more integrative management for species requiring control. Thus, management should be more co-operative and engaging with stakeholders (Shackleton et al., this issue). For example, Palmer et al. (2014) call for alterations to organisational and institutional structures and the building of adaptive capacity to address the negative effects of invasive alien species for livelihoods and human well-being. This included improving education and awareness, building collaborations between institutions and stakeholder groups, engaging and promoting volunteer initiatives, promoting co-management and allowing for flexibility in times of uncertainty. Many of these considerations were also advocated in for the management of *Cenchrus ciliaris* (Marshall et al., 2011). Better understanding of the positions and needs of different cultural and economic groups, and addressing diverse and social values is also crucial and forms a necessary dimension in co-management (Bhattacharyya and Larson, 2014; Ens et al., 2016; Bach et al., this issue). Often initial management approaches are not as successful as initially planned, which highlights need the need for adaptive approaches and flexibility (Cole et al., this issue).

Conclusion

The interplay between invasive species and local livelihoods is highly complex. Some invasive species can be beneficial for some and harmful for others and other species have major detrimental effects which can increase vulnerability within social-ecological systems. Moreover, these relationships and dependencies are not static, but vary through time and with local and broader social-ecological changes. It is therefore important to understand these factors and processes to improve policy and management in the future. This can help with conflict resolution and ensuring that there is evidence to support local and broader-scale decision-making.

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Appendix 1: List of case studies included in the present review

Author/s	Year	Journal
Aravindakshan	2011	Report
Aslan et al.	2009	Society for Range Management
Beinart and Wotshela	2003	Environmental History
Bhattacharyya and Larson	2014	Environmental values
Chikuni et al.	2004	Malawi Journal of Science and technology
de la Fontain	2013	Thesis
de Neergard et al.	2005	Agricultural Systems
Ellender et al.	2010	Water Sa
Engeman et al.	2010	Crop protection
Ens et al.	2016	Biological Conservation
Kannan et al.	2014	Environmental Development and Sustainability
Kaufmann	2004	Ethnology
Kazmi et al.	2009	Report
Keller et al.	2017	Regional Environmental Change
Kull et al.	2007	Mountain Research and Development
Kull et al.	2018	Journal of Environmental Management
Larson	2004	Report
Laxen	2007	Thesis
Llukor et al	2016	Pastoralism
Luizza et al.	2016	Ecology and Society
Marshall et al.	2011	Environmental Science and Policy
Maundu et al.	2009	Biodiversity
McWilliam	2000	Human Ecology
Mosweu et al.	2013	Natural resources
Mungatana and Ahimbisibwe	2012	Natural resources Forum
Mwangi and Swallow	2008	Conservation and Society

Ngorima and Shackleton	2018	Journal of Environmental Management
Opande et al.	2004	Limnologica
Palmer et al.	2014	Human ecology
Pienkowski et al.	2015	Ecological Economics
Poudyal et al.	2017	Human Dimensions of Wildlife
Rai and Scarborough	2015	Small-scale Forestry
Rai et al.	2012	Journal for Nature Conservation
Rijal and Cochard	2016	Regional Environmental Change
Robinson et al.	2005	Conservation biology
Rogers et al.	2017	Pastoralism
Shackleton et al.	2017	Biological Invasions
Shackleton et al.	2007	Human ecology
Shackleton et al.	2015	AMBIO
Shackleton et al.	2017	Biological Invasion
Shackleton et al.	2017	African Journal of Range and Forage Science
Shackleton et al.	2011	Development Southern Africa
Shackleton et al.	2015	Forest Ecosystems
Sheil and Padmanaba	2011	Plant Ecology and Diversity
Siges et al.	2005	Human Ecology
Steele et al.	2006	Journal of forestry
Stronge	undated	Working document
Sundaram et al.	2012	Human ecology
Vaarzon-Morel	2008	Report
Wakie et al.	2016	Applied Geography
Witt et al.	under review	