Land Use Policy 47 (2015) 332–341

Contents lists available at ScienceDirect

Land Use Policy

journal homepage: www.elsevier.com/locate/landusepol

Using social data in strategic environmental assessment to conserve biodiversity

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ARTICLE INFO

Article history:
Received 2 July 2014
Received in revised form 4 March 2015
Accepted 5 April 2015

Keywords:
Strategic environmental assessment
Environmental impact assessment
Biodiversity
Conservation opportunity
Social data

ABSTRACT

Strategic Environmental Assessment (SEA) is increasingly used to assess land use plans in a way that is broader in spatial, temporal and conceptual scope than traditional Environmental Impact Assessment (EIA). Meanwhile, conservation scientists have recognised that successful biodiversity conservation outcomes rely on information about both biological priorities and the feasibility of undertaking conservation actions. SEA provides a framework for integrating information on the social determinants of conservation feasibility with supporting environmental legislation in order to achieve enhanced conservation outcomes. In this paper we argue that data on the social context of land use plans are vital to ensure effective biodiversity conservation outcomes that result from SEAs. We explore the Australian *Environment Protection and Biodiversity Conservation Act (1999)* (EPBC Act) as a case example of how the integration of these data can be practically achieved within an existing legal process. While a range of social data is relevant to this type of assessment, we focus on the use of spatially-referenced social data in the context of land use planning. When applied to the design and implementation of land use plans, this type of information can improve the acceptability of conservation actions, enhance environmental stewardship, and minimise land use conflict through taking stock of the values and attitudes (precursors to behaviour) that are relevant to proposed land use change and conservation action. Through exploring the integration of these data into each of the stages of SEA under the EPBC Act, we show that opportunities exist to strengthen the effectiveness of SEA in delivering conservation outcomes without altering existing legal processes. Yet, for this to be done effectively, practitioners need to be cognisant of a range of theoretical and methodological challenges related to the generation and interpretation of these data, as well as the socio-political context in which they are applied.

Introduction

Assessing the environmental impacts of land use is a standard policy approach of jurisdictions around the world. Environmental Impact Assessment (EIA) is the earliest form of this and is today a tenet of environmental regulation. Since the 1990s, however, Strategic Environmental Assessment (SEA) has increased in prominence (Fundingsland Tetlow and Hanusch, 2012). SEA extends the scope of EIA, moving beyond a focus on isolated actions to also include policies, plans or programs (Partidário, 2000, 1996) and shifts the assessment of impacts to higher orders of decision-making (Fundingsland Tetlow and Hanusch, 2012). For these reasons, SEA has been praised for its ability to consider multiple impacts over much longer time periods and influence the choice of alternative development options rather than simply documenting expected environmental decline (Partidário, 2000, 1996). This is particularly important for biodiversity conservation, as traditional

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http://dx.doi.org/10.1016/j.landusepol.2015.04.002
0264-8377/

individual project assessments have been criticised for their inability to account for cumulative impacts within a larger socio-political context (Partidário, 2000; Slootweg et al., 2001). In contrast to EIA, SEA can “identify threats and opportunities for biodiversity at an earlier stage in the decision-making process” (Treweek et al., 2005, p. 175). Many jurisdictions around the world have therefore adopted elements of SEA as a means of protecting species and environments of national significance that are threatened by large-scale human actions, such as regional plans for urban development or resource extraction (Ng and Obbard, 2005; Uprety, 2005).

Since the 1990s, the field of conservation science has also increased in prominence. This field explores the ecological and socio-economic factors associated with conserving wild nature (Kareiva and Marvier, 2012). Recent conservation science literature has recognised that good outcomes often depend more on favourable social conditions that enable implementation of actions (including human values, attitudes, behaviours and political conditions), than on accurate ecological information (Ban et al., 2013; Carpenter et al., 2009; Knight et al., 2008, 2010, 2006; Knight and Cowling, 2007; Pretty and Smith, 2004; Raymond and Brown, 2011). Much of this research has focused on conservation planning (the identification and prioritisation of areas for conservation) and direct community actions, but has not been used as evidence to support SEA decision making. There is a need therefore to explore how insights from previous research can assist SEA policy officers to make more informed decisions on the social determinants of those outcomes.

Although social and economic factors are increasingly considered within SEA (Morrison-Saunders and Fischer, 2006; Vanclay, 2004), when it comes to evaluating biodiversity impacts, SEA applications around the world remain focused on the physical determinants of environmental damage with little consideration of how social factors might influence conservation outcomes. Treweek et al. (2005) (p. 193) stress that biodiversity impacts “may be influenced by social, economic and political factors” and that these “must be taken into account”. This same sentiment was expressed by the International Association for Impact Assessment (2002), which held that SEA should address the interrelationships between biophysical, social and economic impacts rather than focusing on environmental impacts alone. Relevant data on socio-demographic changes, stakeholder values and behaviour or land use conflicts could help decision-makers identify both opportunities for conservation gains within landscapes, and potential threats that may impede conservation efforts (see Brown and Raymond, 2014).

The widespread use, breadth and inherent flexibility of SEA approaches make for an ideal opportunity to analyse how social data can be systematically considered alongside biophysical data in land use policy. At present there are no standard guidelines regarding the methods that should be used in SEA; each assessment should apply techniques appropriate to the context (Noble et al., 2012). This flexibility is a strength of SEA, yet it can also mean that practitioners are unsure how to gather and implement appropriate social data (Noble et al., 2012). Conservation feasibility refers to the likelihood that an action will be taken that will lead to an effective and sustained conservation outcome (Mills et al., 2013). It is a concept that is increasingly referenced in the conservation literature (Knight et al., 2010; Mills et al., 2013). However, there is currently no guidance on how social data on conservation might be included within impact assessment (Rauschmayer and Risse, 2005). This has implications for the assessment of the social acceptability and feasibility of land-use policies that aim to mitigate or offset the environmental impacts of new developments. We demonstrate here how quantitative measures of social determinants of conservation outcomes can be incorporated into existing methods for SEA, thereby addressing the “need for more systematic methodologies with guidance on methods selection at different SEA tiers and in different contexts” (Noble et al., 2012; p. 145).

In this article, we draw upon the Australian strategic assessment legislation (under the *Environment Protection and Biodiversity Conservation Act 1999 (Cth)*) as a case study of how integrating social data within a statutory SEA approach can enhance conservation outcomes. Since SEAs have been most frequently and successfully applied to land use plans (Fundingsland Tetlow and Hanusch, 2012), we focus our discussion on spatial land use planning assessment, considering in particular how the mapping of social values might enhance SEA in this context. Although the social impacts of plans are important on social justice and democratic grounds (Vanclay, 2003), our concern is specifically how social dynamics might affect conservation outcomes. The emphasis of this article is thus on how to improve the ‘substantive effectiveness’ of SEA (see Chanchitpricha and Bond, 2013), measured by tangible biological outcomes rather than the procedural or transactive outcomes (e.g. improvement of policy process) that have been addressed by other authors (e.g. Sadler, 1996). We begin by reviewing the international literature on the social dimensions of SEA theory and practice. Using the Australian SEA process as a case study, we develop general principles for considering conservation-relevant social data in SEA. We finish by discussing the key lessons from this application and discuss general principles for considering social data in SEA.

Current use of social data in strategic environmental assessment

The definition of SEA is broad and assessment practice varies in form and quality. While SEA in Europe is regulated by the European Union (EU) Directive for SEA (Directive 2001/42/EC), many other jurisdictions around the world lack prescriptive guidance as to how SEA should be conducted. This lack of guidance extends to how the values, attitudes, opinions and behaviours of key stakeholders and the general public should be elicited and incorporated into the SEA process.

Existing methods used in SEAs that explicitly account for social data tend to focus on shared decision-making through participatory approaches (e.g. focus groups) (Gauthier et al., 2011; Rauschmayer and Risse, 2005), which may also contribute to Social Impact Assessment (Vanclay, 2003). Indeed, public participation is widely recognized as vital to effective SEA (Rauschmayer and Risse, 2005), since it provides “transparency and accountability in [the] assessment process” (Noble, 2009; p. 67). Participatory approaches have been applied to conservation issues. For example, collective bargaining of the location of protected areas (e.g. Game et al., 2011) may be an effective way of ensuring successful implementation. However, the focus of these approaches is on the decision-making process and building relationships rather than quantifying social values or preferences for development and conservation for affected communities. This type of participatory planning may not be realistic for large planning regions where there are a large number and diverse range of stakeholders making consensus difficult to negotiate.

We propose that applying quantitative (and often spatially-referenced) social data in an SEA process will enhance biodiversity conservation outcomes in many instances. These types of data can inform the likelihood that biodiversity matters will be threatened as a result of a proposed plan (for example wildlife populations under pressure from increasing nearby urban populations) (Guerrero et al., 2010), or the feasibility of undertaking conservation actions on the landscape (such as establishing a biodiversity offset reserve). Noble et al. (2012), (p. 144) note that although qualitative-based methods of gathering and processing information are often necessary when constrained by short time frames, “[t]here are instances where more quantitative-based methods are perhaps more fitting or where more rigorous methods are warranted". Advocating for the injection of new forms of data into SEA might be seen by some as adopting a 'technical', instrumental or positivist approach to decision-making (Azcarate and Balfors, 2009), rather than the deliberative (political) form of decision-making advocated by many stakeholder engagement commentators (Reed, 2008). However, there are many advantages to applying quantitative social data in this way. First, gathering representative quantitative social data can provide a more democratic community representation in decision-making than approaches that rely on views expressed by a small number of stakeholder groups or individuals (Raymond et al., 2014). Second, decision-making authorities are often more familiar with the process driven analysis typical of EIA. Quantitative social data can therefore be (more) easily integrated into existing legal structures and technocratic decision-making processes. Finally, applying quantitative social data does not preclude participatory approaches to decision-making but can help promote collaborative dialogue between decision-makers and the community—a key element of effective SEA (van Buuren and Nooteboom, 2009). We outline here a novel way of incorporating social data related to conservation outcomes into every stage of an SEA process, thereby enhancing the protection of biodiversity without requiring dramatic transformation of current legislated processes.

Social data relevant to strategic environmental assessment of land use plans

Broadly, social data relevant to SEA of land use plans can be classed as: (1) the individual determinants and indicators of the likely support for conservation such as demographic characteristics, values, preferences, perceived risk, knowledge and access to income support (see Pannell et al., 2006; Raymond and Brown, 2011; Steg and Vlek, 2009; Ticehurst et al., 2011); (2) social interactions (e.g. social networks) and organisational structures that collectively influence biodiversity protection (Guerrero et al., 2013) and (3) the governance systems that influence decision-making and action (Ban et al., 2013; Mills et al., 2013). How and why these data ought to be included in SEA of land use plans are outlined below.

Stakeholder and community values and preferences

The distribution and types of values that individuals assign to places are increasingly relevant to environmental decision-making. Such values are referred to as assigned, social or landscape values (Brown, 1984; Bryan et al., 2011; Seymour et al., 2010). Knowledge of the composition of values for specific locations (such as recreational, aesthetic, or conservation values) can be used to infer relative social importance of these places and the degree of social acceptability of conservation or other land use activities (Brown and Raymond, 2014; Brown and Reed, 2012). These types of data have been used to guide land-use decisions (Brown, 2012) and are known to shape conservation behaviours (Seymour et al., 2010). Land-use or development preference is an additional proxy for conservation feasibility because it reflects a desired end-state or future use of a particular area (e.g. use of land for residential, industrial, or tourism development), which may align with or oppose conservation efforts (Nielsen-Pincus et al., 2010). If local communities prefer development in or near an area of biological importance, the feasibility of future protection of biodiversity in this area is low since conservation will have low social acceptance or political support; in contrast, the feasibility of conservation is high if social values for conservation align with these biologically important areas (Whitehead et al., 2014).

Public Participation GIS is one effective and increasingly utilised method of spatially assessing assigned values and development preferences (Brown, 2012, 2005). Data on spatially referenced landscape values offer four advantages if used together with the biophysical information typically considered in SEAs: (1) identification of the level of compatibility between scientifically assessed conservation areas and areas of local value and concern; (2) prediction of potential conflict zones whereby different types or incommensurable values overlap; (3) allocation of resources to areas of highest biodiversity and community importance and (4) visual representation of the feasibility of plans to protect species of national importance (Raymond and Curtis, 2013). However, there are many theoretical and methodological challenges that need to be considered when using PPGIS as a decision tool. Lechner et al. (2014) discuss many issues related to the spatial accuracy and uncertainty of these types of data, while other researchers have highlighted other issues that can impact data quality such as spatial discounting, the influence of survey design, and decisions around which spatial attributes on which to focus (Brown and Kyttä, 2014).

Social networks and organisational structures

Social Network Analysis (SNA) is increasingly applied in a wide range of contexts including public health, business, and natural resource management. Recently it has also been used in a number of conservation applications (e.g. Guerrero et al., 2013; Mills et al., 2014). SNA essentially documents the number, type and strength of connections between different individuals (nodes) in a particular context. A node's position within a network is thus considered to influence its opportunities and constraints, which in-turn affects the outcomes it can bring about (Ban et al., 2013). The benefits of including SNA data in the strategic assessment of land use plans are that it allows for the identification of (i) existing collaborations between key stakeholder groups in the region of interest, (ii) key individuals that need to be on side for a land use plan to be successfully implemented, (iii) key stakeholder groups that new bridging relationships need to be built between and (iv) the degree of disconnection in the social networks in the area of interest. Depending on the stakeholders involved, strong or weak connections between parts of a social network can influence the ease in which a land use plan can be implemented successfully.

Governance systems

Governance systems are understood as a fundamental component of social-ecological systems and have potential to greatly affect conservation outcomes (Ostrom, 2007). Key elements of governance include agencies, their structures and interactions, legal systems, social norms, rules and policy systems (Ban et al., 2013). In the context of SEA of land use plans, it is important to know how conservation areas or assets will be governed over time. The influence of local governance on biodiversity outcomes was highlighted in a recent study of marine resources in Melanesia (Mills et al., 2013). There, traditional self-governance arrangements such as temporary closures of fisheries or species restrictions were highly correlated with perceived fishery conditions. Any SEA that was to be conducted in this context would need to take account of these governance arrangements and cultural practices. A different governance system would likely yield quite different outcomes for biodiversity. Weak governance of biodiversity assets (either formal or informal) may make their conservation difficult to implement, even if specified within a plan undergoing a SEA.

Case study: opportunities for applying social data in strategic environmental assessment in Australia

We consider here how quantitative social data can be applied to SEA by using Australia as a case study. We begin by outlining the function of SEA in Australia, before exploring how it can be adapted to account for the social determinants of biodiversity conservation success. Despite a policy focus on strategic consideration of long-term scenarios and participatory decision-making among stakeholders, application of SEA in Australia, along with many other jurisdictions, is tethered closely to EIA philosophy and is motivated by legal requirements to report on specific impacts (see Lobos and Partidario, 2014). The legal weight behind SEA in Australia is the *Environment Protection and Biodiversity Conservation Act 1999 (Cth)* (EPBC Act). It provides for both single project focus EIA (Parts 7, 8 and 9) and the broader approach of SEA (Part 10) called a 'strategic assessment'. This type of assessment consists of a voluntary agreement between the Australian Government and a project partner. Under the EPBC Act, the delegated Government Minister has ultimate power to approve or reject actions or classes of actions relevant to a development proposal likely to have a significant impact on Matters of National Environmental Significance (MNES) (such as threatened species and ecological communities). To date, there have been 18 voluntary strategic assessments conducted under the EPBC Act. Approval to proceed with actions or a class of actions relating to a development proposal has been granted to a range of assessment partners, including national offshore petroleum activities, urban expansion and water supply infrastructure development.

The EPBC Act's strategic assessment provisions differ from EIA in that they consider the impacts on MNES from a *series* of proposals or developments across larger temporal and spatial scales, rather than an individual project (DSEWPAC, 2012). This can permit development across a larger area without further need for individual project assessments (DSEWPAC, 2012; Early, 2008). This approach has been viewed favourably by many developers who considered that site-based assessments of threatened species 'stifled development' and economic growth, and resulted in substantial transaction costs, often referred to as 'green tape' (see Macintosh, 2012; Debret, 2012 for an overview of the arguments). The potential environmental benefits of such an approach are apparent in its coordinated nature. While a standard EIA considers specific site based impacts, the strategic assessment considers impacts at a landscape scale, thus reducing the risk of multiple impacts that can deteriorate the social and ecological integrity of a landscape system. However, recent commentaries have queried whether these broader scale assessments are sufficiently rigorous to protect biodiversity (Tager, 2013). Consequently, there has been a formal House of Representativeness Committee enquiry into "Streamlining environmental regulation, 'green tape' and one stop shops" (Commonwealth of Australia, 2014). One stop shops refer to a single point of processing for developers to address provisions under state and national environmental regulation or legislation.

Application of SEA in Australia is interesting in the international context for two reasons. First, the location of strategic assessment provisions within the EPBC Act means that the impact significance of a proposal is measured entirely against impacts to MNES, although the Minister must consider economic and social factors related to the proposed action (Macintosh, 2009). This narrow focus on environmental concerns is similar to New Zealand practice but differs from the UK and many European countries that typically place greater weight on broader sustainability concerns (Jones et al., 2005). Second, there is no legal requirement for a strategic assessment to be undertaken. This differs from most European countries where an SEA is mandatory for land use plans under the European Union Strategic Environmental Assessment Directive. However, the steps that constitute a strategic assessment in Australia are formalised and more highly regulated than the flexible approach taken in other countries (e.g. Canada) (Fundingsland Tetlow & Hanusch, 2012; Jones et al., 2005).

Under the EPBC Act, strategic assessments are undertaken within the broad framework of a standard SEA process (UNEP, 2002). These stages are outlined in Table 1 and summarised graphically in Fig. 1.

Rarely is there explicit discussion between the consent authority and assessment partner about the social determinants of conservation within a strategic assessment. In this context, we outline how social data can be used within the eight stages of strategic assessment (depicted in Table 1) and why it is important for enhancing conservation outcomes. Although these insights are directed towards SEA in Australia, they are relevant to many

Table 1
The stages of the strategic assessment under the EPBC Act.

Stage	Description
1. Screening	The screening stage is used to assess whether a particular policy, plan or programme should be subject to strategic assessment (Early, 2008), based on a pre-determined set of criteria for identifying likely significant impacts on MNES.
2. Scoping	The scoping stage is undertaken collaboratively to negotiate a formal agreement between the Minister and the assessment partner as well as the terms of reference for the assessment process (Marsden, 2013). This stage identifies important issues, how to examine them, and which guidelines to reference (DSEWPAC, 2012).
3. Impact analysis and assessment	Once the scale of impact of the policy, plan or program has been determined, the assessment partner and consent authority collaboratively look for ways to reduce the identified impacts to acceptable levels. This could include avoidance, mitigation or compensatory actions, such as environmental offsets (Macintosh, 2013)
4. Consideration of mitigation measures	Should the impact assessment stage conclude that the plan, policy or programme is likely to have significant environmental impacts, this mitigation measures stage is designed to look for ways to reduce the identified impacts to acceptable levels. This could include avoidance, mitigation or compensatory actions.
5. Reporting	The reporting stage has three main priorities: (i) to document the findings of the assessment, the proposed alternatives and predicted impacts, (ii) to serve as a basis for consultation, and (iii) to provide recommendations for decision-makers, based on preferred alternatives and measures for avoiding, minimising, mitigating and compensating for unavoidable impacts. This process includes comments from the public and any advice from the consent authority.
6. Review	This stage is designed to act as a check on the adequacy of the information collected as part of the strategic assessment process, including identification of bias, uncertainties and contradictory findings. Once finalised, endorsement and feedback from the Minister is sought.
7. Decision making	Following consideration of the matters raised in the strategic assessment, the Minister may approve the taking of actions, allowing activities under the policy, plan or program to proceed without the need for further federal approval of individual development proposals (Ashe and Marsden, 2011). However, conditions may be attached to an approval if the Minister considers them necessary. Critically, any decision must also take account of any relevant economic and social matters of the plan, policy or program (EPBC Act, s146F).
8. Monitoring and environmental auditing	Monitoring and auditing is conducted by the assessment partner in relation to the mitigation measures agreed to with the consent authority. This takes place beyond the decision-making stage to ensure that the protection of MNES is upheld throughout the life of the strategic assessment agreement (DSEWPAC, 2012). This can include monitoring both social and ecological change and the performance of agreed mitigation measures.

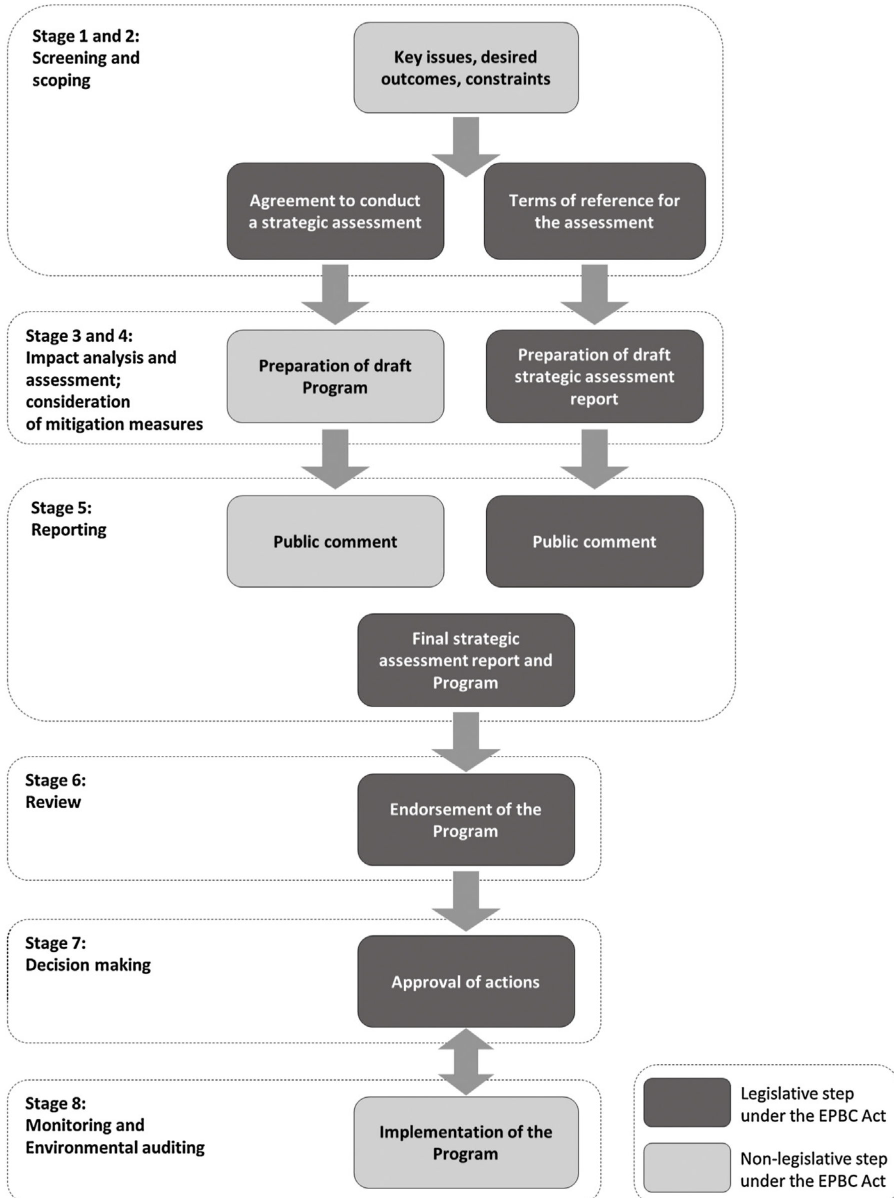

Fig. 1. The strategic assessment process under the EPBC Act (adapted from DSEWPAC, 2012).

applications internationally, as the stages represent standard SEA practice around the world (UNEP, 2002).

Screening and scoping (stages 1 and 2)

At present, social investigation within the strategic assessment process is generally limited to expert consultation and engagement pertaining to the physical requirements of particular MNES, with relatively little emphasis on broader community values. The screening and scoping phases could be enhanced by utilising data on how social behaviours, attitudes, values and priorities relate to MNES, the proposed development and its anticipated environmental impacts (e.g. Curtis et al., 2005). For example, mail based surveys can provide baseline contextual information for drafting, negotiating and progressing the strategic assessment terms of reference (see Raymond and Curtis, 2013 for example). Moreover, understanding how people value and use species (e.g. for fishing) and habitats (e.g. wilderness recreation) is a critical first step to identifying socially meaningful conservation priorities within an area (Ives and Kendal, 2014). Without this information, it is possible that key issues may be overlooked, which may undermine the efficacy of the remaining strategic assessment process.

During the initial stages of a strategic assessment it is important that the role of social data is discussed amongst stakeholders and policy professionals, and practitioners are well versed in conservation feasibility literature and practice. This will require effective communication between social scientists and policy-makers, which has long been recognised as a challenge (Vaughan and Buss, 1998). The science-policy divide can in part be bridged through intentional 'boundary work' on the part of social scientists (Rose, 2014) (defined as ways in which scientists "construct, negotiate, and defend the boundary between science and policy"; Owens et al., 2006, p. 640). Strong governance arrangements are also needed to incorporate this level of complexity within the impact analysis process. A poor investment in understanding social values at the initial stages of the assessment process can lead to public grievances in later stages, including legal proceedings, protests and public campaigns. These have the potential to delay or even stall complex projects (Franks et al., 2010).

Impact analysis and assessment—consideration of mitigation measures (Stages 3 and 4)

During this phase social data can provide a benchmark against the terms of reference to identify community values and activities that are either beneficial or detrimental to protection of nationally protected species. For example, areas of ethno-biological significance, traditional hunting value, scenic quality, recreational importance and social well-being may relate positively to the protection of MNES, and should feature in the assessment report. Similarly, certain land use preferences, recreational activities, employment types and resource uses may conflict with conservation outcomes. Data can be collected via maps of aboriginal cultural landscapes (Ridges, 2006), visitor perceptions of park experiences, environmental impacts, and facilities (Brown and Weber, 2011), social values for natural capital and perceived threats (Bryan et al., 2011), and willingness of landholders to steward natural resources (Pasquini et al., 2010).

The assessment of impacts stemming from a proposed plan should consider indirect changes to biodiversity resulting from alteration of the social factors discussed above. For example, shifting demographic profiles arising from proposed development (e.g. an increase in residential density or the number of young families present within a region) could change how people interact with areas of significant biodiversity, such as regional parks. Also, disruption of land management regimes (e.g. hunting or fishing behaviours) can lead to ecological degradation, even though the development associated with a proposal itself may not influence habitat directly. In terms of mitigation and offsetting of impacts, social data such as willingness to sell for conservation (Guerrero et al., 2010) and willingness to pay for environmental improvements (Brouwer et al., 2010), can also assist in developing options that will be biologically favourable and socially sustainable.

Public consultation and reporting stage (stage 5)

Public consultation can be modified to include evaluation of the accuracy and adequacy of the social data (collected at stages 1–4). Participatory mapping and modelling methods can also be used to facilitate community engagement, accounting for the needs of multiple individuals or groups of individuals (Lesslie, 2012; Voinov and Bousquet, 2010). The visualisation of impacts through mapping data is particularly useful for this purpose. A key consideration at this point is to ensure appropriate representation of relevant communities and to avoid only engaging a biased group (Brown, 2012).

Review and decision-making (stages 6 and 7)

Social information can inform the endorsement decision and the application of any necessary approval conditions. For example, an approval condition for a development impacting a threatened ecological community might include capacity building for the establishment of an Indigenous peoples bush foods industry, thereby creating a synergy between economic development, species protection and social licence to operate. However, many proponents may not be experienced in this kind of activity, and may require additional support and guidance to ensure compliance is met. Data on socio-ecological linkages can also assist assessment officers and the assessment partner to develop governance mechanisms under which impact management would be best implemented.

Monitoring and environmental auditing stage (stage 8)

In addition to direct monitoring of legally protected matters, there is potential for ongoing evaluation of the social factors (individual or collective) that may indirectly influence their persistence. For example, understanding the management capacity of local councils or nature reserve staff can provide assurance to the Minister that conservation outcomes for threatened species will be achieved. Finally, social data can be used to broadly assess the outcomes of Natural Resource Management (NRM) instruments used for avoiding, mitigating and offsetting environmental impacts (e.g. Curtis et al., 2008) and provide lessons for refining current and future strategic assessments.

General principles for considering conservation-relevant social data in SEA

A number of planning and policy principles can be derived from our study of the strategic assessment process in Australia that relate to SEA applications globally. There is great variation in SEA legislation, methodologies and procedures internationally and it is beyond the scope of this paper to review these here (but see Fundingsland Tetlow and Hanusch, 2012 for a discussion). Nevertheless, whether or not SEA is perceived as a rational way of evaluating environmental impacts or a loosely implemented framework for developing collaborative sustainability solutions (c.f. Fundingsland Tetlow and Hanusch, 2012), most SEA contexts will contain opportunities to integrate social data in the assessment of conservation outcomes.

A stepwise approach to considering key social matters related to biodiversity in SEA practice

A number of logical steps should be followed by both the parties preparing reports to be assessed and those performing an assessment. First, it is important that SEA practitioners "consider biodiversity values and uses within the plan area" (Treweek et al., 2005, p. 188). Once relevant biodiversity matters are identified (either on social or biological grounds), the social determinants of conservation within the landscape need to be considered. The next consideration is then to understand how relevant social conditions are likely to change with the implementation of a plan.

Assessment of threats and opportunities for conservation that are associated with a policy, plan or program is perhaps the most significant stage within an SEA. This can be done by considering three landscape categories. The first is existing protected areas, which are the cornerstone of most conservation efforts. Questions that should be asked include (i) are they likely to persist in providing conservation outcomes into the future?, (ii) what is the current and likely future level of social acceptability? and (iii) how threatened are they by shifting community attitudes and changing behaviours? The second landscape category is biodiversity outside of formal protected area networks. Questions to be asked of these areas include (i) what social capital (Pretty and Smith, 2004) exists to maintain and enhance biodiversity on private land? and (ii) how might this change with the implementation of the policy, plan or program? If a large proportion of the biodiversity being considered under a SEA is present on private land, answers to such questions may be crucial to conservation outcomes. The final landscape category is newly created protected areas. This is becoming increasingly important with the rapid adoption of biodiversity offsetting in SEA. The capacity of new conservation reserves to meet biodiversity outcomes is dependent to a large degree on their design, management and political and community acceptability. Moreover, creation of formal reserves as offsets may not lead to better biodiversity outcomes as this shift in land tenure may promote abdication of responsibility by landholders. An understanding of community attachment and stewardship is very useful in determining where to position such biodiversity offset areas.

Operational guidance

One key recommendation for effective integration of social data with environmental data in the SEA process is that both should be collected concurrently throughout the data collection stages as a requirement of the assessment partner. The kind of social data collected will depend on the context of the plan, with secondary data collection (e.g., review of grey and peer-reviewed literatures) possibly sufficient in communities frequently surveyed by social scientists. However, the use of public participation techniques to elicit social values (such as PPGIS) has the added advantage of achieving other outcomes than simply enhancing biodiversity protection. These include learning outcomes (both social and technical), governance outcomes (such as enhancing stakeholder participation in decision-making), development outcomes (influencing the design of plans), and attitudinal and value changes (promoting sustainability within the community) (Fundingsland Tetlow and Hanusch, 2012). The analysis of social and environmental data together can also help identify socio-ecological tipping points, where activities undertaken can cause phase changes to natural and social systems. Consideration of such complex concepts will require the collaboration of interdisciplinary teams of practitioners and the integration of conservation and social impact reports. Effective integration of social data collated using instrumental, quantitative assessments with deliberative valuation processes will also require careful reconciliation of different perspectives on rationality, processes of value elicitation, types of representation and degree of involvement of decision-makers (Raymond et al., 2014).

SEA practitioners should also look for existing opportunities in legal structures for the inclusion of social data related to conservation outcomes, as this article has demonstrated for the Australian context. Indeed, a robust analysis of social values provides decision makers with increased certainty that decisions regarding protection of environmental assets are more legally defendable. Since the overarching purpose and language of SEA is broad and inclusive of environmental, social and economics elements of sustainability, most frameworks for the application of SEA contain relevant clauses or operational practices that can support the inclusion of these data.

Potential challenges to the application of social data in strategic environmental assessment

Although social matters are critical to achieving conservation success, there are a number of challenges that could affect the application of social information to Strategic Environmental Assessment of land use plans. While our case study focused on the Australian strategic assessment process, the challenges discussed here are relevant to SEA applications more broadly.

Data collection and integration

The cost of data collection can pose an economic challenge to the use of social data in a strategic assessment. Mail-based surveys are costly and time-consuming compared with the collection of secondary data, such as that from publically available census databases. However, mail-based surveys enable a targeted assessment of community attitudes toward particular issues related to biodiversity conservation, such as the impact of regional demographic change and property turnover on the adoption of natural resource management practices by landholders and the future viability of agricultural industries (Mendham and Curtis, 2010; Mendham et al., 2012). Regional census data only allows for extrapolations of the impact of developments on socio-demographic trends.

One way to overcome the cost of social data collection is to interpolate self-reported social impacts in the study area from known biophysical characteristics in related regions (Sherrouse et al., 2011). Spatial interpolation techniques are based on known correlations between biophysical features (e.g., vegetation cover, species distribution) and social data (e.g., attitudes toward residential development, local values for conservation). However, projecting social data spatially and applying data from one region to another carries with it many assumptions which introduce uncertainty and error in analysis (Eicher and Brewer, 2001; Gotway and Young, 2002). An alternative is for multiple development and environmental agencies to work together at the sub-regional or regional scale to collect social data within a consistent methodology. However, there are well-known challenges to inter-agency collaboration that would need to be overcome. Ranade and Hudson (2003) cite five categories of barriers to collaboration that would apply to the coordination of social data collection. They are (1) structural; e.g. organisational complexity or fragmentation of responsibilities, (2) procedural; differences in planning horizons and cycles, (3) financial; e.g. differences in the availability and flow of resources, (4) professional; e.g. differences in ideologies and values or conflicting views about user interests and roles and (5) status and legitimacy; e.g. organisational self-interest (Ranade and Hudson, 2003).

Collection of social data should account for the differing effect of social dynamics according to the scale of analysis; the biodiversity of landscapes, catchments and properties will all have different social drivers. It is also important for decision-makers to be cognisant of the fact that social drivers of conservation can often operate at different scales than the biodiversity outcomes (Cumming et al., 2006). Furthermore, some social issues may not have been revealed via regional survey methods, and planning agencies may need to undertake more detailed analysis in areas where developments are likely to have the highest social and/or environmental impact. Some of these cross-scale issues can be overcome through adopting multi-level environmental governance frameworks which foster coordination and collaboration across multiple tiers of government (Newig and Koontz, 2013; Prager et al., 2015, 2012). Specifically, they could be used to link social data collected as part of municipal surveys to social data collected through sub-regional or regional surveys.

It can be challenging to assess how strongly social matters influence biodiversity because of the complexity of individual and group processes (Pannell and Vanclay, 2011). Strategic Environmental Assessments of land use plans may therefore need to make greater allowance for the complex associations between social values, attitudes, behaviours and environmental outcomes, rather than rely on proven causal relationships (Biggs et al., 2011; Johnson et al., 2013). This would provide a stronger role for self-reports of attitude, impacts and risks in the assessment process. In the case of Australia, there is a risk, however, that focusing too much on social data (that are often only indirectly associated with environmental outcomes) could expose the consent authority to legal challenge, since the Government's legislated authority extends only to the protection of MNES. We therefore do not argue that these social data should necessarily be given equal weight as biophysical factors, but rather that their influence be applied systematically in context of such factors.

Organisational implementation

The culture of proponent organisations and regulatory authorities is likely to influence how successfully social data are incorporated into the SEA processes. Organisations that are used to dealing predominantly with biophysical information can perceive that social information is less useful for decision-making because it is 'soft' or imprecise (see for example Bojórquez-Tapia et al., 2003). Resistance to the use of social data in assessing biodiversity impacts may need to be combatted by addressing this perception (Brechin et al., 2002; Robertson and Hull, 2001). Good leadership and providing avenues for civil servants and proponents to express any concerns can be proactive ways of bringing about cultural change. The commitment to change literature suggests that change is most likely to occur when employees are aware of a change, have the skills to implement it, are motivated to support the change through adequate incentives and share the vision driving the change (see Jaros, 2010 for a review). As most SEA practitioners are used to evaluating biophysical impacts of a proposal, there may be a lack of skills and expertise in integrating these with relevant social data. This could result in misinterpretation of social data as it relates to biodiversity impacts, or the neglect of useful social information altogether. This can be addressed through targeted training for both proponents and assessment staff on (i) what kinds of social data are relevant for different assessments, (ii) methods on collecting social data and (iii) how to interpret social data as it relates to conservation outcomes.

Stakeholder engagement can be another potential challenge to successfully integrating social data into SEA. Stakeholder engagement can be difficult for many reasons, including different mental models and perspectives held by alternative stakeholder groups (Biggs et al., 2011), legal and institutional barriers to stakeholder collaboration, and unequal power relations among stakeholders. To best navigate these challenges, the scope and purpose of the engagement need to be articulated clearly to stakeholders at the outset of the project to ensure that societal expectations regarding data use are accurate. Following data collection, translation of social data relevant to the assessment to stakeholders and general public must be done carefully, with clear communication about the implications of the information. If social data are not made accessible and understandable to stakeholders and decision-makers they are unlikely to influence the decision-making process (Biggs et al., 2011; Knight et al., 2011).

Finally, decision outcomes may not reflect the new information even if social data are integrated well into reports and documents that inform an SEA. Macintosh (2013), (p. 542) notes that improved information alone may not generate better environmental decisions in EIA, since decisions are largely the product of "values, power and incentives". This is a key issue that is currently being grappled with by scholars. Indeed, Pope et al. (2013, p. 7) note that the responsibility for impact assessment is commonly allocated to "government agencies whose mandate is development rather than those whose mandate is environmental protection". Social data can be selectively reported to support a pro-development agenda and, in the case of SEA, may be used to dilute environmental outcomes (Morrison-Saunders and Fischer, 2006). This is an issue that is not easily resolved and is in need of further research. However, the use of representative quantitative social surveys are perhaps more robust to political distortion than other social data collection techniques.

Conclusion

Incorporating social determinants of conservation success in SEAs can strengthen conservation outcomes. Failure to do so can lead to unforseen negative biodiversity impacts following changes in social dynamics that result from actions undertaken according to policies, plans or programs. SEA offers great promise because of its widespread use, broad scope (considering more diffuse upstream causes of environmental impacts) and flexible administration. Recognising that the primary role of conservation legislation is to provide environmental protection, novel solutions are needed for utilising this legislation to explore the social determinants of such protection. Our case study of the Australian strategic assessment process demonstrates that opportunities exist within current legal processes for seamless integration of this data to enable improved conservation outcomes without altering or compromising the current legislative arrangements. Since it is widely accepted that successful conservation relies on the feasibility of undertaking conservation actions, legal mechanisms providing protection for biodiversity cannot afford to neglect the relationships between social and natural systems and should be updated to reflect this, both for the sake of long-term environmental conservation and the integrity of the legislation. However, many questions remain about the practical application of social data to SEA, and the role of power and governance structures in influencing the effective use of this information to achieve conservation outcomes. As these academic questions continue to be pursued, stronger continued collaboration between conservation scientists and environmental regulators will be required to advance the contribution of social data to strengthen conservation outcomes in legislated SEA processes both in Australia and internationally.

Acknowledgements

This article is a product from a workshop on the theme of conservation opportunity, held on Stradbroke Island, Queensland, 23–26 April 2013. Funding for the workshop and resulting research is from the Australian Government's National Environment Research Program, and the Australian Research Council Centre of Excellence for Environmental Decisions. The authors would like to thank Steve Mercer and Ross Rowe for their constructive feedback on an earlier version of this manuscript and two anonymous reviewers whose comments have substantially improved this manuscript.

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