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Mitigation for one & all: An integrated framework for mitigation of development impacts on biodiversity and ecosystem services

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A B S T R A C T

Emerging development policies and lending standards call for consideration of ecosystem services when mitigating impacts from development, yet little guidance exists to inform this process. Here we propose a comprehensive framework for advancing both biodiversity and ecosystem service mitigation. We have clarified a means for choosing representative ecosystem service targets alongside biodiversity targets, identified servicesheds as a useful spatial unit for assessing ecosystem service avoidance, impact, and offset options, and discuss methods for consistent calculation of biodiversity and ecosystem service mitigation ratios. We emphasize the need to move away from area- and habitat-based assessment methods for both biodiversity and ecosystem services towards functional assessments at landscape or seascape scales. Such comprehensive assessments more accurately reflect cumulative impacts and variation in environmental quality, social needs and value preferences. The integrated framework builds on the experience of biodiversity mitigation while addressing the unique opportunities and challenges presented by ecosystem service mitigation. These advances contribute to growing potential for economic development planning and execution that will minimize impacts on nature and maximize human wellbeing.

1. Introduction

Governments, health organizations, aid agencies, and more recently, conservation organizations, have goals to improve the lives of people through development that also preserves the life support systems of the planet. Simultaneously achieving these goals is challenging and nearly all countries have approached this dilemma by creating legal and policy requirements for mitigating the environmental impacts of development (Morgan, 2012). Impact mitigation frameworks applied by many governments and lending institutions around the world are consistent in their strong support for the mitigation hierarchy, which involves first evaluating whether avoiding and minimizing these impacts are possible, and where not feasible or sufficient, offsetting or compensating for residual effects (Lawrence, 2003; McKenney and Kiesecker, 2010). The stakes for implementing strategic development goals are especially high: the rate at which energy, water, and infrastructure development projects are growing is accelerating with total investments expected to exceed $53 trillion between 2010 and 2030 (OECD, 2012).

To help inform impact mitigation, the scientific community has responded with decades of research establishing best practices for applying the mitigation hierarchy to biodiversity impacts (Race and Fonseca, 1996; Geneletti, 2002; Landis, 2003; BenDor et al., 2008; Canter and Ross, 2010; BBOP, 2012b). Despite these efforts, the approach has fallen short in practice for both biodiversity and the benefits it provides to society—ecosystem goods and services (collectively referred to as ecosystem services, or ES, for simplicity). Minimizing and offsetting impacts on biodiversity and ecosystem function have been the primary focus of mitigation historically, but such efforts can fail to avoid impacts on critical habitats (Clare et al., 2011), often do not account for cumulative impacts at a landscape scale (Canter and Ross, 2010; Kiesecker et al., 2010), inconsistently and inadequately account for ecological equivalency in losses and gains (Quétier and Lavorel, 2011) and seldom succeed in returning lost biodiversity and ecosystem function (Zedler and Kercher, 2005; Maron et al., 2012). These shortcomings largely stem from a historic approach to mitigation that is reactive, with actions focused at small spatial scales and on a project-by-project basis.

To address these shortcomings, biodiversity mitigation policies and programs are now moving away from site-based, piecemeal mitigation to a scale that can more comprehensively account for cumulative impacts of development within a region (Saenz et al., 2013a,

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2013b, Villarroya et al., 2014) and even at a national scale (Kormos et al., 2014). There is general consensus now among researchers and practitioners that biodiversity and ecosystem function mitigation should consider whole systems, anticipate impacts, and proactively recommend compensatory actions (Kiesecker et al., 2010; Hayes, 2014). This larger-scale approach is supported by researchers and practitioners and is expected to more accurately capture ecological dynamics, and allow for more strategic and proactive mitigation planning. Instead of simply requiring replacement of impacted resources in similar sites in close proximity to the impacts, compensatory mitigation can be steered to priority areas for both ecological and socio-economic investment, likely resulting in better outcomes (Wilkinson et al., 2009).

At the same time that improvements in biodiversity mitigation have been recognized and solutions put forth, there is growing recognition that ecosystem services have largely been forgotten (Brownlie et al., 2012; Bos et al., 2014). Ironically, many of the laws that establish mitigation requirements were designed to protect people from environmental degradation associated with development: in other words, to guard against ecosystem service loss (Villarroya et al., 2014). The language in these laws ranges from the general to the specific. For example, Australia's National Strategy for Ecologically Sustainable Development is designed to "enable development that improves the total quality of life, both now and in the future, in a way that maintains the ecological processes on which life depends". In much more detail, the U.S. Clean Water Act §404 that establishes the foundation for wetland and stream mitigation states that "management programs shall conserve such [clean] waters for the protection and propagation of fish and aquatic life and wildlife, recreational purposes, and the withdrawal of such waters for public water supply, agricultural, industrial and other purposes".

In addition to these legal precedents, there is a growing demand for ecosystem service impact assessment and mitigation by international organizations and multi-lateral lending agencies. For example, the Organization for Economic Co-operation and Development (OECD) has developed guidance for addressing ecosystem services in Strategic Environmental Assessment (SEA) (OECD, 2008). Within the financial sector, the Performance Standards of the International Finance Corporation (IFC) now require that projects they finance adhere to the mitigation hierarchy for both biodiversity *and* ecosystem service impacts (IFC, 2012). Current implementation, however, does not meet the intent of these laws and new standards. For example, in the U.S., wetlands damaged by development in urban centers are being mitigated for in more rural areas with lower population densities. Even if these mitigation actions meet biodiversity mitigation needs, they will still fail to return wetland-related ecosystem service benefits to the people who have lost them (BenDor et al., 2008).

2. An integrated framework for biodiversity and ecosystem service mitigation

Although suggestions have been made for how to include biodiversity or ecosystem services separately for specific kinds of assessments (e.g., SEAs, Geneletti, 2011) and in specific contexts (e.g., Kiesecker et al., 2010; Tallis and Wolny, 2010), a systematic and unified approach for integrating services with biodiversity into the mitigation hierarchy is lacking. To address this gap, we build on previous work to propose an integrated framework that allows regulators to determine potential, cumulative impacts on biodiversity and ecosystem services (BES) at a landscape, watershed, or seascape scale and to assess the compatibility of development with environmental and social goals. Our recommendations stem from decades of research on best practices for mitigating development impacts on biodiversity in terrestrial landscapes, which are relevant for and can be tailored to freshwater and marine systems (Bos et al., 2014). The framework addresses development siting, impact estimation and offset assessment, which are all iterative steps in an adaptive assessment and mitigation process (Fig. 1).

Clearly this integrated treatment of BES in mitigation is challenging given that BES are unique, non-interchangeable, and determined by related, but often different environmental factors. As such, there are few places in the mitigation hierarchy where the same data, analytical processes, and activities can be applied consistently for both components. Here we review the current state of the art for biodiversity mitigation and compare and contrast biodiversity approaches with the conceptual challenges of ecosystem service mitigation. We discuss each step of the mitigation hierarchy in detail below, highlighting potential BES synergies and outstanding research needs with the goal of advancing integrated best practices for impact mitigation that more holistically account for people and nature.

2.1. Siting

In the first phase, development options (both individual and suites of projects) would best be placed within a landscape or seascape context to guide their appropriate siting. Targets are selected, the spatial extent is determined, and conservation plans (Fig. 1A) can be used to capture potential cumulative impacts and guide the selection and avoidance of development sites.

2.1.1. Selection of targets

Although a comprehensive consideration of BES would be ideal (Geneletti, 2011; IFC, 2012), data and resource limitations will ultimately restrict the number of species, habitats, and ES that can be considered in impact mitigation assessments. Despite such constraints, it is important to recognize that biodiversity and ES are not interchangeable, either across their own respective elements or across groups. A woodpecker is not the same ecologically or in terms of social value as a leopard, and water for irrigation is not the same as crop pollination. Beyond this obvious statement of uniqueness, BES often exhibit different spatial and temporal patterns, and so should not be considered as consistent surrogates for each other (e.g. Egoh et al., 2008; Naidoo et al., 2008; Cardinale et al., 2012). Given the hundreds to thousands of options for targets to use in a BES impact assessment, and the fact that the choice of targets greatly determines the outcomes of mitigation (Eiswerth and Haney, 2001), a systematic selection procedure is needed to ensure that the subset of BES targets chosen is as representative as possible.

Biodiversity targets should be selected based on their ability to approximate the complete biological diversity of a region or site and to indicate key changes in ecological conditions due to predicted local or landscape-scale changes including development impacts and climate change. Common approaches for selecting adequately representative biodiversity targets have been reviewed and discussed extensively (Margules and Pressey, 2000; Poiani et al., 2000; Groves et al., 2002; Kiesecker et al., 2009) (Fig. 2a). In practice, mitigation tends to focus on sites and species with protected status (e.g. nature reserves, Sites of Special Scientific Interest, IUCN Red List taxa), on economically important game species or charismatic species, or on at-risk habitats and species (e.g., rare, threatened, or endemic species). Greater adherence to existing recommendations, such as a focus on multiple 'umbrella species' that span different development threat categories (Roberge and Angelstam, 2004), will better capture the full suite of biodiversity impacts in the development region (Geneletti, 2002; Gontier et al., 2006).

As with biodiversity, we face a major challenge in effectively representing the diverse set of ES provided in any given area. Provisioning services, such as food production, water supply and timber production, are over-represented in research and data collection (Millennium Ecosystem Assessment, 2005; Russell et al., 2013). The selection of representative targets in the ES realm can be achieved in part by considering a suite of services that fall under the broad categories of provisioning, regulating, and cultural services as defined by the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2005; Fig. 2b). Not all ES will be relevant in all development contexts, but consideration of all categories will help to ensure

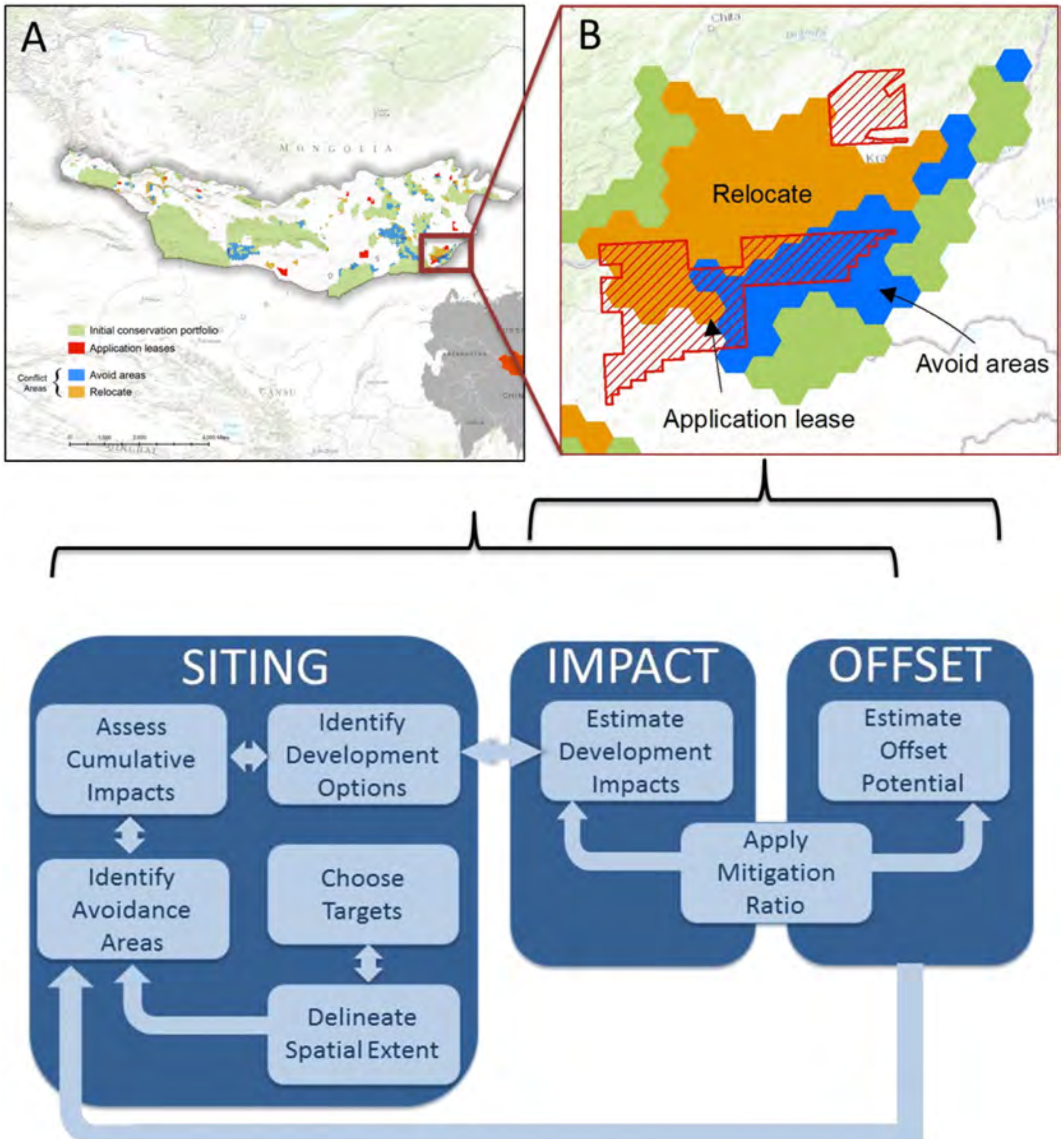

Fig. 1. Integrated framework for biodiversity and ecosystem service (BES) mitigation for development impacts. Siting would be done in the context of a landscape- or seascape-scale assessment so that areas critical for BES priorities are avoided and cumulative impacts can be considered. Targets are the species, habitats, genes, and ecosystem goods or services prioritized in any region or case. A landscape-scale assessment of critical habitat avoidance areas in Mongolia's Gobi Desert shows the landscape (A) and project (B) view of biodiversity avoidance and ecologically equivalent offsite sites. For more details see Heiner et al., 2013.

that major benefits are not overlooked. Assessors may consider the use of stakeholder engagement approaches to help identify locally preferred or important services (e.g., Rosenthal et al., 2014), though Agency or Ministry mandates and regulatory context will also feature heavily in the selection of focal ES. Best practice suggestions developed for the United States federal government offer useful considerations for choosing focal ES (Olander et al., 2015).

Within ES categories, further effort will be needed to select a representative subset of targets. Both the number and characteristics of beneficiaries (or people who receive a specific service) are important considerations. Services impacting a larger proportion of the affected population or particularly vulnerable cohorts (e.g., poor, women, elderly, indigenous groups) may be prioritized. Additionally, services could be targeted if they are expected to suffer the greatest losses from development to ensure that the most.

significant social or ecological damages are avoided or at least accounted for. Targets may represent services with a disproportionate amount of their supply within the proposed development zone (Luck et al., 2012), or services that are highly sensitive to proposed development activities. Target services may also be chosen if they have high social or commercial value for their continued delivery (e.g. access to spiritually significant gathering places, commodity food production, high-value timber production) (Luck et al., 2012).

Expanding the 'umbrella species' concept from biodiversity theory, ES whose provision corresponds well with a myriad of others may be used as 'umbrella services' (Daily, 2000), to presumably capture the trends and potential losses of other non-target services. For example, regulating services were found to be highly correlated with the most diverse set of services in Quebec, Canada, indicating that they could serve as proxies for other services in this region (Raudsepp-Hearne et al., 2010). However, careful consideration of local conditions must be used when choosing service targets in this way, as inter-relationships are often unknown and can vary with biophysical conditions, land use, and socio-political context, among other factors (Anderson et al., 2009; Nelson et al., 2009; Cardinale et al., 2012). For example, in southwest Australia, water quality regulation and carbon sequestration are positively correlated, making either service representative of the other, but the two are negatively correlated in the Argentinian pampas making them poor

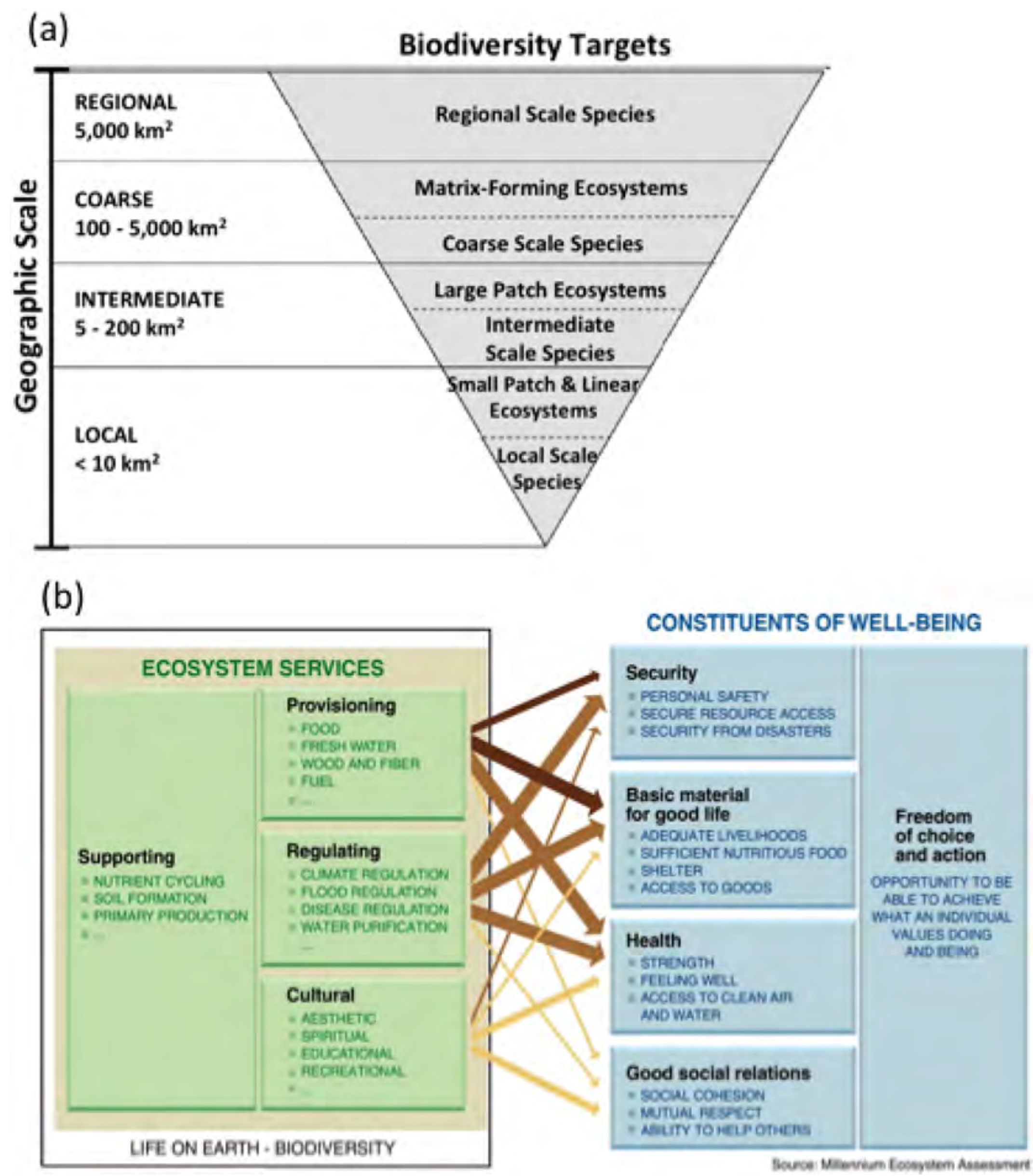

Fig. 2. Frameworks for biodiversity (a) and ecosystem service (b) target selection. Priorities for impact mitigation should be selected to ensure representation of non-target species, habitats, or ecosystem goods and services. Representation across spatial scales and levels of biological organization from local to regional has been advocated in biodiversity conservation planning (reprinted Poiani et al., 2000, with permission). For ecosystem goods and services, we suggest representation across types of services as defined in the Millennium Ecosystem Assessment (reprinted from Millennium Ecosystem Assessment, 2005).

surrogates there (Jackson et al., 2005). These correlations also may not be stable over time as climate and other global drivers change ecosystem functions and inter-relationships between ES.

In addition to these general criteria for ES selection, concepts for the selection of biodiversity targets are relevant in cases where the production of ES is tightly tied to specific species or habitat types. For example, commercial and recreational fishing services and value are usually dominated by a few socially preferred species (Holmlund and Hammer, 1999). Carbon storage and sequestration benefits are tightly associated with vegetation communities, which can be characterized relatively well by habitat or land cover types (Gibbs et al., 2007). Moreover, the provisioning of some ES is correlated with the biodiversity of a system (Cardinale et al., 2012). When relationships between services and species are known, positive, and sufficiently strong, biodiversity target selection criteria can be applied to services as well.

2.1.2. Delineate spatial extent

Spatial boundaries for impact mitigation are conventionally attributed to the local impact site(s) and any reference or control sites if evaluated (Atkinson et al., 2000; Gontier et al., 2006). Expansion in scope from this single project-scale to a larger, landscape-scale has been recognized as essential for cumulative impact assessments of multiple stressors in complex socio-ecological systems (Margules and Pressey, 2000; Geneletti, 2002; Groves et al., 2002; Landis, 2003; Landis and Wiegers, 2007). We further suggest that to effectively mitigate BES impacts of development, this scale should capture not only the estimated extent of influence(s) for the proposed development activities at hand, but also the net extent of current and forecasted future development (as done in Copeland et al., 2009).

In order to capture the distributions and dynamic processes supporting biodiversity targets across that larger landscape or seascape scale, the spatial extent needs to coincide with ecoregional or watershed boundaries and allow for assessments of habitat function or condition, species' population viabilities, community assemblages, and larger-scale processes like fragmentation (Gontier et al., 2006; Groves et al., 2002). Overall, in order to maintain the collection of biodiversity targets, their spatial extent should include joint boundaries that capture the spatial units for all targets (as explained above) as well as the ecological processes that support them.

The core spatial unit for ES targets is the area that can provide the same benefits to the same people, termed the 'serviceshed' (Fig. 3) (Tallis and Polasky, 2009). The spatial extent of a serviceshed is determined by the area that supports biophysical service production, and allows beneficiaries both physical and institutional access to the service (Tallis and Polasky, 2009). If there is no biophysical supply of a service (e.g. no water purification taking place, no fish that can be caught, no forest views to enjoy), then there is no benefit. If formal (laws, regulations) or informal (social norms, cultural practices) institutions restrict beneficiaries' ability to access the biophysical supply, then there is no benefit. Institutions that limit or promote access can be very diverse, and can include protected areas, irrigation rights, land tenure, traditional rights, hunting or fishing seasonal closures, and many others. Institutions may also be variably applied to different groups of people, emphasizing the need to map out such institutions to accurately account for impacts on or potential mitigation benefits for specific groups of people. Finally, if people cannot physically access services that require such access then no human benefits can accrue. If clean water is flowing in a river, and no institutions limit access to it, but there are no pipes, roads or paths to allow access, then no one may be able to garner use benefits from that clean water. Physical access also often varies among social groups (e.g. wealthier people may access water via delivery pipes to their homes while poorer people may use footpaths to wells or rivers), again emphasizing that these spatial relationships must be clearly defined for impact assessment and mitigation.

All three of these elements (biophysical supply, institutional access, physical access) can apply to use and non-use services alike. However, serviceshed boundaries, and the importance of each factor in determining them, can vary dramatically among services. Some services, such as carbon sequestration, encompass the planet, as the atmosphere is well mixed (so supply anywhere provides benefit to all people equally) and no institutions or physical barriers prevent any people from benefitting. Existence value is another service where physical and institutional accesses are not key in defining the serviceshed, as existence is enjoyed non-physically. In other words, beneficiaries do not need to be in contact with a species to enjoy it, and so there are no physical or institutional limits to the serviceshed boundary. As such, the serviceshed for existence value of a given species is simply defined as the habitat range for that species (or the area of biophysical supply for that species' population).

Physical and institutional constraints can be more relevant for more locally provided services, such as drinking water quality or esthetic value. Both require people to be able to physically and legally access the supply (e.g. the clean water or a pleasing view) for the benefit to flow. The watershed upstream from an extraction or use point captures the drinking water quality serviceshed, as it captures the area supporting water quality and regulation processes, and can be drawn as appropriate to reflect both institutional and physical accesses. To extend this example, if source watershed areas include inter-basin diversions, then the serviceshed incorporates all pertinent upstream contributing areas above all water diversion infrastructure (McDonald et al., 2014). If a city relies on water piped in from a reservoir 50 km away, that city's drinking water regulation serviceshed is the watershed upstream of the reservoir, not the watershed upstream of the city itself.

The serviceshed for crop pollination (Fig. 3a) would be the area around a pollinator-dependent crop field within the flight distance(s) of local, wild pollinators in relation to habitats (foraging and nesting resources) in a landscape. Habitats that lie outside the flight range of pollinators would house those pollinators, but would not be providing a pollination service. The serviceshed for water quality regulation to support recreational fisheries (Fig. 3b) captures the upstream catchments from lakes 1) containing desirable recreational fish species, 2) within a tolerable driving distance of beneficiaries, 3) where recreational fishing is legally and/or culturally allowed, and 4) where physical access is possible (e.g. boat launches, piers or fishing access points exist) (Fig. 3b). Lakes too far away, without access, without desirable fish, or with restricted fishing rights would not fall within this serviceshed boundary.

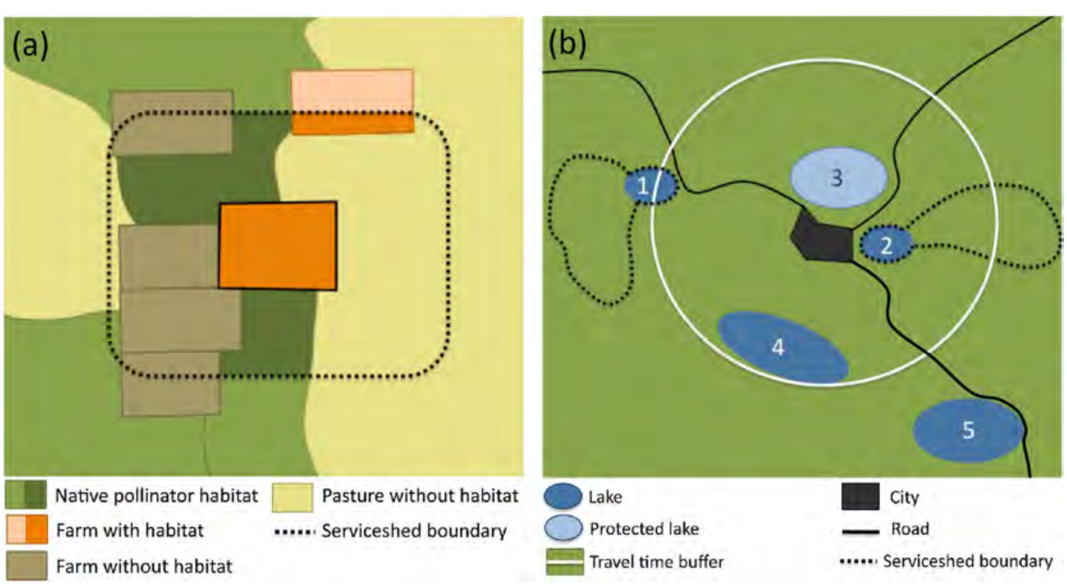

Fig. 3. Hypothetical serviceshed boundaries. Serviceshed boundaries for crop pollination (a) are determined by the flight distance(s) of pollinators in relation to crop fields and habitat areas. Darker colored farm and native habitat are within the serviceshed area, while lighter habitats are not. Pollinators may live outside the serviceshed boundary but will not provide pollination service to the focal farm. The serviceshed for recreational fisheries (b) is determined by the lakes (or rivers) with harvestable recreational fish species that are within an acceptable travel time and that allow physical access (public access, boat launches, etc.). Lakes 4 and 5 are outside the example serviceshed because potential beneficiaries lack physical access or are too far away, respectively. Lake 3 is within the potential serviceshed area but is protected, so access is denied due to institutional restrictions.

Other proposed definitions for the spatial units of ES have focused largely on the ecological entry points and have been based on ecosystems (Hein et al., 2006) or service providers (Luck et al., 2009a, 2009b). These definitions scale ES appropriately to their ecological production areas, but fall short by not including elements like infrastructure, access or restriction zones, locations of markets, and other social and economic factors that determine the amount and value of services enjoyed by people (Boyd and Wainger, 2003; Guerry et al., 2015). A relevant assessment unit for services would allow for clear estimation of any distributional effects, accounting for factors like these among others. A consequence of avoiding these social distributional factors is illustrated as described above with the US Clean Water Act (Ruhl and Salzman, 2006).

Serviceshed boundaries can be delineated for any prioritized service and related beneficiaries just as boundaries for biodiversity can be proxied by ranges, occurrences, densities, numbers, and interactions of species and habitats (Luck et al., 2009a, 2009b; Tallis et al., 2012a; Mandle and Tallis, 2012; Arkema et al., 2013). Ultimately, the spatial extent of impact assessment and mitigation should be defined as the fullest extent needed to capture both biodiversity (and associated ecological processes) and the servicesheds (and associated beneficiaries) of all services targeted.

2.1.3. Avoidance criteria

Once the targets and boundaries are set for mitigation analysis, determination of how and where to avoid impacts is the first formal step in the mitigation hierarchy. This step is meant to avoid creating negative environmental and social impacts from the outset through careful design and placement of development activities (Nichols and McElfish, 2009). Here, we focus on the spatial aspect of avoidance: the locations where the risks of development are likely to be too high for mitigation efforts to achieve a no net loss (or ideally net gain) in biodiversity or ecosystem services (BBOP, 2012b, Gardner et al., 2013, McKenney and Kiesecker, 2010). Although information, guidance, and tools exist to inform the identification of avoidance areas for biodiversity, they are seldom applied in practice (Clare et al., 2011), leading to a net loss of species, critical habitats, and functioning ecosystems (Race and Fonseca, 1996).

Priorities and thresholds for impact are often based on two key factors: irreplaceability (or uniqueness) and vulnerability (or threat) (Margules and Pressey, 2000; BBOP, 2012a; Pilgrim et al., 2013). In a biodiversity mitigation context, irreplaceability refers to the spatial extent of options that will be lost if a particular site is impacted by development. Vulnerability refers to the likelihood that mitigation actions will return the same biodiversity benefits as those that were impacted given current losses and impending threats. Several general factors determine the irreplaceability and vulnerability of biodiversity in any given mitigation context (Fig. 4).

The irreplaceability of biodiversity in a site typically is assessed based on the prevalence of rare, unique, endemic or geographically restricted species or habitats, the extent of remaining intact and undisturbed critical habitats, and the importance of an area in supporting key source populations and/or evolutionary processes (e.g., key migratory routes or unique genetic diversity) (Fig. 4) (BBOP, 2012a; IFC, 2012; Margules and Pressey, 2000). In simple terms, the irreplaceability of an area increases with the percentage of the global range or overall population of biodiversity target(s) it contains (Langhammer, 2007; Pilgrim et al., 2013).

Ecosystem service avoidance has not been addressed systematically in the mitigation literature although the relevance of irreplaceability and vulnerability has been recognized (Brownlie et al., 2012; Gardner et al., 2013). We extend the same concepts from biodiversity mitigation guidelines to ES and propose that several factors are likelyto affect the availability of alternative areas for mitigation of services if a given site is impacted (Fig. 4).

General Factor	Biodiversity	Ecosystem Services	AVOID if...
Landscape-level BES Goals	Options to protect/mitigate target species or habitats, or services within servicesheds		Few
	Multiple values overlap for BES		Yes
Irreplaceability (uniqueness)	Intact, undisturbed or critical habitat [1]		Yes
	Endemic, rare, unique species or habitats		Yes
	Protected areas		Yes
	Support for ecological connectivity or evolutionary processes		Strong
		ES link to specific place, habitat, or species	Strong
		Beneficiary dependence on ES	Strong
		Size of serviceshed area	Small
Vulnerability (threat)	Current rate and extent of loss from other drivers		High
	Rate and likelihood of recovery from disturbance		Low
	Highly threatened species or habitats		Yes
		Beneficiary access to alternatives	Few
		Expected future ES demand	High

[1] One definition of critical habitat is offered by the IFC Performance Standard Number 6 (IFC 2012): "Critical habitats are areas with high biodiversity value, including (i) habitat of significant importance to Critically Endangered and/or Endangered species; (ii) habitat of significant importance to endemic and/or restricted-range species; (iii) habitat supporting globally significant concentrations of migratory species and/or congregatory species; (iv) highly threatened and/or unique ecosystems; and/or (v) areas associated with key evolutionary processes." While this definition encompasses other factors listed in the table (i.e., endangered or endemic species, ecological connectivity, evolutionary processes), we still list them since definitions for critical habitat could differ across policy contexts.

Fig. 4. Factors influencing avoidance for mitigation of biodiversity and ecosystem service (BES) impacts. Avoidance areas should be defined as those where impacts cannot be mitigated effectively and equivalently elsewhere. Conditions where mitigation will likely be difficult (all else equal) are identified in the "AVOID if…" box. Factors pertinent only to biodiversity are shown in green, for ecosystem services in blue, and overlapping considerations in orange. These factors are offered as general guidelines, but landscape- and site-level quantitative assessments are needed to identify specific avoidance areas (or thresholds) for a given region.

Irreplaceability will be higher in areas where target ES are linked with specific species or habitat types than in those dominated by service targets that can flow from a broader range of species or habitats. For example, a site harboring a rare fish species used in a traditional ritual practice for a localized indigenous tribe will be more difficult to replace than a site where the largest service benefit stems from carbon storage (since the latter can be provided, to varying degrees, from a multitude of habitat types at any location on the globe). Similarly, services provided within smaller servicesheds are likely to be more irreplaceable because they are reliant on spatially constrained variables, thereby reducing the mitigation options to conserve or restore other service provision areas within the same serviceshed. For example, water-related ES that depend on catchment-scale hydrologic functions must be mitigated within the same catchment (serviceshed) to provide the same benefits to the same people (Tallis and Wolny, 2010).

The vulnerability of biodiversity at a site is determined in part by how quickly and reliably target species or habitats can recover from disturbances given their sensitivity and exposure to threats and their current (background) rates of loss (Fig. 4) (BBOP, 2012a; IFC, 2012; Wilson et al., 2005). Already threatened or at-risk species and habitats are common examples of highly vulnerable biodiversity targets (Langhammer, 2007; Pilgrim et al., 2013). This same rationale holds for ES, such that sites with higher rates of services loss or that support services with slower or less likely recovery trajectories should be avoided for development. The level of vulnerability is also influenced by the ability of beneficiaries to respond to service loss and their access to substitutes. Beneficiaries are likely to be less vulnerable to loss if the ES provided in an area are not directly contributing to basic needs or strongly held values. In addition, if people have access to viable and affordable alternatives for the same services, either naturally or technologically, they will be less vulnerable to the loss of those services. Holding all else constant, we propose that avoidance is more appropriate where environmental impacts occur to places providing ES directly linked to basic human needs and strongly held values, and where few alternatives exist.

Ideally, the irreplaceability and vulnerability of sites in relation to proposed *and* cumulative development projects would be assessed at each site within a broader, landscape context (Gardner et al., 2013). As the impacts on any given site and its role in maintaining BES varies based on local context and socio-environmental conditions, universal thresholds for avoidance factors do not exist. Therefore well-informed decisions are based on project-specific assessments within a regional context that include population and habitat viability, habitat fragmentation and connectivity, and ES assessments in conjunction with scenario analyses informed by landscape-level conservation and development planning exercises. When practical limitations prohibit such detailed assessments, practitioners often base decisions on the best available science and on best judgment about the likely risk of development. Standards have been established to identify important areas for biodiversity that include both the occurrence of threatened species (e.g. according to IUCN Red List) and certain percentages of species' populations (Langhammer, 2007; Ricketts et al., 2005). These thresholds have also been proposed as criteria to determine critical areas that should be avoided for development (IFC, 2012; Pilgrim et al., 2013).

Equivalent decision criteria with a focus on ES attributes and how they respond to human induced pressures are only in the early stages of development. Concepts have been proposed to identify ES thresholds, such as the "safe minimum standard" or the minimum quantity of ecosystem structure and process …that is required to maintain a well-functioning ecosystem capable of supplying services" (p. 2053 in Fisher et al., 2008). The goal of these efforts is to establish a minimum conversion benchmark that ensures sustainable service provision (analogous to minimum viable population) (Kontogianni et al., 2010). But in reality, these thresholds are difficult to validate or put into practice given the uncertainty around predicting future ES demand and the fact that BES interactions are likely to influence resilience (Bennett et al., 2009).

Given that our understanding of development impacts on BES is quite limited for a wide variety of human land uses and activities (Alkemade et al., 2009; Carpenter et al., 2009; de Groot et al., 2010), and that there are capacity constraints to conduct regular and consistent assessments for development proposals (Morgan, 2012), the most risk-averse approach is to proactively conduct landscape-scale assessments of candidate avoidance areas. Once avoidance areas are identified, they can be given systematic consideration as new projects are proposed and can be used to inform development options in the further stages of the mitigation hierarchy.

2.2. Impact

Proposed development, when compatible with landscape-level conservation plans, will have the least impact if it follows best management practices to minimize impacts at the site and to mitigate for the unavoidable impacts using strategic compensatory actions. During the impact stage, projected impacts are assessed and efforts to minimize, restore and offset impacts are considered.

2.2.1. Minimize and reduce impacts

Minimization seeks to reduce the duration, intensity, and/or extent of impacts that cannot be completely avoided. Once a site is chosen for development, impacts can be minimized by reducing the size of area impacted, relocating or focusing site activities in less sensitive areas, and/or adopting less damaging activities within the site's active areas. While the factors that determine sensitivity vary, all of these approaches are applicable for both biodiversity and ES minimization. On-site minimization represents the most widely used approach for biodiversity, with design and engineering solutions the most active area of practice (The Energy and Biodiversity Initiative, 2004). After attempts to avoid or minimize impacts are made, measures can be taken to rehabilitate or restore degraded ecosystems or reclaim cleared land post-impacts. Like minimization, restoration techniques for biodiversity impacts are an active area of research and practice (The Energy and Biodiversity Initiative, 2004).

Too little work has focused on ES minimization to identify leading practices. However, much can be gleaned from techniques developed for related processes such as water quality regulation, water supply, air pollution and flooding (Nichols and McElfish, 2009). We also suggest that ES may benefit more so than biodiversity from on-site activities that use engineered options to minimize impacts. For example, water quality regulation may be impacted on a site where surface vegetation is cleared and soil is disturbed. Built water filtration and treatment facilities could replace these benefits to the broader water supply system, and in so doing, reduce off-site service losses. Air quality regulation, soil fertility, fish production, carbon sequestration and coastal protection are among the other ES that may benefit from on- or near-site technological alternatives, though this area has received little research attention to date.

2.2.2. Impact assessment

Once development plans are adjusted to reduce and minimize impacts on-site, the residual impacts of activities can be estimated. Many quantitative tools exist to estimate the potential development impacts on biodiversity. Functional assessments are ideal, and include methods for assessing the condition and functioning of habitats like wetlands (Kusler, 2003) and biological and ecological models for assessing species and community persistence (Gontier et al., 2006). Ideally, impact is assessed over the projected lifetime of the project, and other land or sea-scape trajectories are considered such that estimated impact takes into account likely future development or other (e.g. climate) changes at broader scales that influence the degree of impact that will be experienced at a given site. These methods have been reviewed elsewhere (Geneletti, 2002).

Despite guidance for best practice on biodiversity impact assessments, common practice fails to implement the state of the art tools and methods (Gontier et al., 2006), most commonly failing to estimate cumulative impacts and estimating impacts on the basis of area alone. Area-based impact assessment considers impacts equivalent in all acres of a given habitat type (e.g. wetland) regardless of the quality or functioning of the habitat or the services it provides (e.g. in retaining species diversity or providing sediment and nutrient retention benefits for water quality). This area-based assessment approach has been widely criticized for its inability to adequately compensate for development impacts (Ruhl and Salzman, 2006; Wilkinson et al., 2009). In response, functional assessments have been recommended as a means to more effectively capture ecological variability and function (McKenney and Kiesecker, 2010; Quétier and Lavorel, 2011).

2.2.2.1. Basic principles for ES impact assessment. An equivalent conceptual basis for ES impact assessment does not yet exist, but relevant methods have been advanced in several other contexts. In general, it is recognized that ES can be assessed for their *potential* to benefit people (supply), the amount of service *actually used or enjoyed* by people (delivery), or for people's *preference* for receiving the amount of service (value) (Tallis and Polasky, 2009; Granek et al., 2010).

In a mitigation context, rigorous functional ES impact assessments focus not only on how supply is disrupted but also on how the delivery of services will be impacted over the lifetime of the project, and how much those changes are likely to matter to people (value). Ideal ES impact assessments calculate impacts of a given project in the context of other land- or sea-scape scale changes likely to happen over the lifetime of the project, such that they capture cumulative effects of development and other changes (e.g. climate) along with human population and other changes that alter the demand for and value of a given ecosystem service. Estimates of impact on supply alone will miss the important connection to people, and as such, do not accurately reflect an impact on people's ability to receive and benefit from an ES. For example, development that reduces surface water flows reduces the potential supply of drinking water, but if no people are using those surface water flows for drinking, then there is no impact on service delivery. Ideally, impact assessments seek to focus on changes in ES value, but value data are often more limited than biophysical data on service delivery, and a strict focus on service value may un-necessarily restrict the set of services that can be assessed (Tallis et al., 2012b). As such, ES impact assessments will at best estimate impact on value, and at least include an assessment of impact on ES delivery.

For services that are provided by a single species or habitat type, functional biodiversity impact assessment methods can be reliably used to estimate impacts on service supply. However, even for these services, additional methods are needed to assess impacts on service delivery or value. For example, population (or habitat) viability analysis may indicate how development will impact pollinator populations, but such an analysis does not indicate how much pollination *service* is likely to change. In this context, the service-provider unit (SPU) concept can be useful for impact assessment (Luck et al., 2009b). When there is a tight link between services and particular species, as in the case of pollination, it makes sense to link the appropriate measure of biodiversity (population density, functional diversity, etc.) and the levels of service delivery. However, these clear links between species identity or diversity exist for only a subset of services, and have been described for an even smaller subset of services (Cardinale et al., 2012). For example, carbon sequestration, water quality regulation, flood mitigation and air pollution regulation are among the many services that can be provided to differing degrees by a diverse set of species and habitat types.

In all cases, ES impacts are best estimated as the marginal change in ES delivery or value to each beneficiary (Tallis and Polasky, 2009). Marginal change is the difference between ES delivery or value under baseline conditions and under proposed development scenarios. Simply assessing the current level of service delivery or value is insufficient because it is unlikely that the total service amount will be lost as the result of land conversion or degradation, and such methods could inadvertently overestimate the impacts of development. It is also important to estimate the marginal impacts on each target service and beneficiary independently. As discussed above, services do not always have the same relationships with each other (e.g. tradeoffs or synergies) or to human benefits, so any use of one service as a proxy for another must be established as valid in any given context.

2.2.2.2. Methods for ES impact assessment. Given these requirements, some methods and tools developed for biodiversity assessment are appropriate for ES impact assessment, while others are not. Area-based or habitat-based assessments have limited utility (Olander et al., 2015), as they seldom allow for the estimation of marginal impacts or of the impacts on individual services. They instead commonly focus on 'total ecosystem value' and rely on the lumping of services that may not accurately reflect the set of services flowing from the focal analysis area. They also commonly assume a consistent delivery or value of service from a given area (unit) of habitat; for many services, the amount of service provided varies within habitats as a function of biophysical conditions, demand, institutions and physical access that limits or allows services to be realized (Barbier et al., 2008; Koch et al., 2009; Ricketts et al., 2008). Tools and databases focused on benefits transfer approaches (e.g. Natural Assets Information System™, Wildlife Habitat Benefits Estimation Toolkit, parts of Toolkit for Ecosystem Service Site-based Assessment) often use area-based and habitat-based studies, so any application of values from these sources must be done with care to extract service-specific, marginal change information matched to the study site (Plummer, 2009).

The ideal method for ES impact assessment is to use a functional assessment (e.g. Diaz et al., 2011), preferably in the form of ecological production functions to estimate ES flows and to directly tie them to beneficiaries are ideal for impact assessment (Slootweg et al., 2001; Boyd and Wainger, 2003; NRC, 2005; Olander et al., 2015). These methods use equations that capture key factors and relationships that allow for the calculation of marginal impacts and the estimation of service-by-service impacts, as well as reflect the variability in ecological and socio-economic conditions both among sites and over time.

Several tools have been developed on this basis (ARIES, InVEST, MEASURES, MIMES), and their strengths and weaknesses have been reviewed elsewhere (Vigerstol and Aukema, 2011; Bagstad et al., 2013; Waage et al., 2011). Data limitations are often a challenge in the use of production function approaches, and several available tools have overcome these limitations by using land use/land cover (LULC) data as a key input to ES models (Nelson et al., 2009; Raudsepp-Hearne et al., 2010). This differs from a simpler habitat-based approach (e.g. x services provided by y acres of wetland habitat) because LULC data are combined with other data that reflect key variability in service provision within a LULC type (e.g. soils, elevation, climate, management data), and connects these ecological factors to the location and intensity of human demand for the service (e.g. via travel time models for recreation, or infrastructure access points for drinking water).

A production function approach is currently limited for many services by an absence of critical secondary data, high resource or capacity demands for primary data collection, and/or the absence of practical production function-based assessment models (but see Kareiva et al., 2011; Guerry et al., 2015). These challenges have led some to suggest the use of ES indicators in other decision contexts (e.g. de Groot et al., 2010 for land use planning) and some efforts have been made to compile lists of such indicators. However attractive the use of proxies or indicators may be, their utility for creating marginal impact estimates is tenuous at best, and the relationships between services and their potential proxies remain largely untested for most services (Naidoo et al., 2008).

2.3. Offset

Even when development is consistent with landscape-scale conservation goals, and best practices are used to minimize impacts, some impacts are unavoidable. Offsets can take the form of positive management interventions such as restoration of degraded habitat, or protecting areas where there is imminent or projected loss (BBOP, 2012a). For BES, offset design methods need to ensure that the targeted actions provide additional replacement for unavoidable negative impacts, involve measurable, equivalent gains and target effective and efficient placement. To address these issues we suggest a two step process, where 1) the potential benefits of mitigation actions are estimated across a large area, and 2) offset sites are chosen based on that potential to efficiently meet the mitigation requirement (Kiesecker et al., 2009; Tallis and Wolny, 2010).

2.3.1. Estimate offset potential

Estimating the additionality – or benefit – of an offset is the conceptual inverse of impact assessment, and as such, the same principles used in impact assessment apply to offset potential estimation. In this step, the intent is to estimate the marginal improvements in BES expected from identified mitigation actions. For internal consistency, the same methods need to be used for both impact and offset assessment. As with impact assessment, guidance for estimating ES offset potential is less developed, but the principles outlined above hold. Area- and habitat-based methods are not sufficient for capturing the likely differences in ecological condition associated with biodiversity targets, the change in ES attributable to mitigation actions, nor for capturing variation in those benefits associated with differences in ecological and social conditions. Functional assessments again will provide the most robust and informative estimates of how offset benefits are likely to vary across land- and seascapes, and will be most effective if applied to estimate the marginal gains between a scenario where development proceeds without offsets and one where offsets are implemented.

2.3.2. Offset design

Once offset potential has been estimated across the assessment region, locations can be selected that meet the mitigation requirements. Many resources have been developed to guide biodiversity offset design specifically (BBOP, 2012b) and other sources of information can be adapted for this use (e.g. ecoregional assessments, watershed management plans and other conservation planning exercises (Kiesecker et al., 2009; Wilkinson et al. 2009; Kiesecker et al., 2010; Weber and Allen, 2010)). Despite the availability of information to guide offset design, the use of biodiversity offsets in practice remains limited (Salzman and Ruhl, 2005; McKenney and Kiesecker, 2010) and most biodiversity offsetting that is done is focused on-site. This narrow focus on activities near or at the site of impact means that opportunities on the broader land- or seascape that may contribute more to overall biodiversity condition are missed. In other words, the best place for mitigation on the impact site may have much less ecological potential or be much less cost-effective at returning the lost biodiversity than activities at a site elsewhere on the landscape. For both biodiversity and ecosystem services, we strongly recommend a comprehensive land- or seascape-scale approach to find the most effective mitigation opportunities. In fact areas identified for avoidance in many cases will serve as the best offset options if identified thru a landscape-scale assessment accounting for cumulative impacts (Kiesecker et al., 2009; Kiesecker et al., 2010).

2.3.2.1. In-kind vs. out-of-kind offsets. For biodiversity offsets, most policies include a requirement for like-for-like or in-kind offsets: those that conserve similar attributes of biodiversity to those affected by the development (Salzman and Ruhl, 2005; McKenney and Kiesecker, 2010). There are some situations in which better conservation results may be obtained by placing the offset in an ecosystem of higher conservation priority than that affected by the development. For example, limitations on the availability of offset sites that can provide in-kind mitigation can lead to the allowance of a different approach called out-of-kind mitigation where impacts on one biodiversity target are allowed to be replaced by improvements in a different biodiversity target (Bull et al., 2013). In addition a regional landscape perspective may provide opportunities to identify situations in which “trading up” offsets offer valuable alternatives that deliver better conservation outcomes from out-of-kind offsets. Whether in-kind or out-of-kind mitigation is allowed affects how offset sites are selected.

Guidance on ES offsets is in early stages of development (Brownlie et al., 2012). We offer three key considerations: whether interchange among services will be allowed (e.g. water quality regulation impacts offset by pollination benefits), whether interchange between beneficiaries will be allowed (e.g. impacts to one city replaced by benefits to another city, or impacts to indigenous people offset by benefits to non-indigenous people), and whether monetary or other compensation for lost services will be allowed (e.g. payments for lost services or technological provision of an alternative such as bottled water for lost water filtration services). Ultimately, what is allowed is likely to be dictated by political preference. However, it must be clearly recognized that offsets that provide different services than those lost in development or that focus on different beneficiaries than those impacted will, by definition, create ecosystem service ‘winners’ and ‘losers’, and very likely create or deepen social inequities with regard to service delivery. This potential has been demonstrated through the Clean Water Act in the United States, where failing to require in-kind offsets of ES to the same beneficiaries has led to the redistribution of wetland-related benefits from low-income city dwellers to relatively higher income rural dwellers (BenDor et al., 2008).

2.3.3. Spatially combining biodiversity and ecosystem service offsets

With a clear view of how offset benefits for BES vary over space, and knowledge of whether in-kind or out-of-kind options are allowed, practitioners can then identify which offset sites are likely to meet offset requirements most efficiently. Relatively sophisticated prioritization approaches exist for biodiversity targets, building heavily on conservation planning practice (Groves et al., 2002; Possingham et al., 2000; Kiesecker et al., 2009, 2010) and these same methods can be adapted to information identifying how ES benefits vary across space (Chan et al., 2006).

Any prioritization exercise will need to follow a logical order for choosing BES offsets. There are three options for how prioritization could address these two sets of targets. In one option, biodiversity offsets could be prioritized first, using classic methods to identify the suite of sites that most efficiently meets biodiversity mitigation requirements. Then, that set of sites could be assessed for the benefits it returns to ES. If the biodiversity offset sites do not return sufficient ES benefits, a second round of prioritization could be done to identify the best sites to fill in the remaining ES offset requirements. This method was followed in a case where ES considerations were added to existing biodiversity regulations in Colombia (Tallis and Wolny, 2010). In another option, this approach could be reversed such that ES offsets are prioritized first, choosing sites that efficiently meet the ES offset need. These sites could then be assessed for biodiversity benefits, and a second prioritization used to fill in additional area needed to meet unfilled biodiversity offset requirements.

The third option is the most truly integrated prioritization, in which biodiversity and ecosystem service offsets are considered in a joint prioritization. In this approach, sites would be chosen based on their ability to meet *both* biodiversity and ecosystem service offsets. This method was applied to identify areas to conserve habitat in the Brazilian Cerrado to bring lands into compliance with the Forest Code in a way that benefits both biodiversity (bird and mammal species) and water-related ecosystem services (nutrient and sediment retention). In this case, a third objective, minimizing agricultural opportunity cost, was also included in the prioritization to help increase the feasibility of identified, optimal offset options (Kennedy et al., in review).

In contexts where biodiversity and ecosystem service targets have high spatial correlation, any of the above spatial prioritization options will give similar results, and selection of a method is less significant. In areas where there is little spatial overlap among targets, choice of method will strongly influence the selection of sites, and will require serious consideration. For example, a mitigation assessment of a proposed road through the Peruvian Amazon compared how different offset options affected indigenous and non-indigenous communities in the impact region, emphasizing how offset placement would affect social equity (Mandle and Tallis, 2012). In this case, ecologically driven offsets with ES fill in were compared to jointly targeted offsets for several water quality regulation services. Neither approach could fully compensate for lost ES benefits, but including services information in the spatial prioritization reduced projected residual impacts to drinking water quality more than 4-fold for sediment loads, 16-fold for nitrogen pollution and nearly 40-fold for phosphorous pollution. Given the minimal availability of such studies, research is insufficient to identify which method is most informative or efficient under different ecological and social conditions.

In addition to scientific considerations of where offsets for BES can be most efficiently sited, policy rules will be important in defining the degree to which it is desirable to promote overlapping BES offsets. This issue relates to the "stacking" debate in the ES literature regarding whether policymakers should allow or prevent compensation for spatially overlapping services as separate units, each compensating for different impacts (Robertson et al., 2014). In designing mitigation policies, policymakers need to determine whether to allow stacking, a decision that may influence whether offset activities deliver a credible environmental benefit without unintended of un-assessed ES.

2.3.4. Mitigation replacement ratios

Offset policies generally seek no-net-loss or net-gain outcomes for conservation (McKenney and Kiesecker, 2010; Quétier and Lavorel, 2011; Gardner et al., 2013). For these outcomes to be achieved, practitioners need to develop a framework that estimates how much an offset project compensates for project impacts and to help identify which offsets maximize conservation return by delivering the highest-value conservation at the lowest cost and risk. Under existing policies, offset benefits are often estimated using mitigation replacement ratios, which establish the number of credit units that must be created by an offset action to compensate or replace one unit of loss at the project site. The most common current practice is to define offsets in habitat area units. For example, a ratio of 4:1 would mean that 100 impacted hectares of habitat would need to be offset by 400 ha of the same habitat elsewhere.

There are no standard practices for establishing mitigation ratios, and ratios vary dramatically. Common values are reported below 10:1, reaching higher ratios in some cases (Moilanen et al., 2009; McKenney and Kiesecker, 2010; Saenz et al., 2013a). Ratios are often negotiated, inconsistently representing the many factors that are likely to determine the actual match between impact levels and offsets. Both biodiversity and ecosystem service ratio calculations would benefit from a consistent approach that considers a standard set of variables, and matches the importance and relevance of those variables to the context and the assessment methods used.

Based on a literature review of commonly considered offset factors, we introduce a factor set that can serve as the starting point for a standard approach to BES ratio calculation (Table 1). Factors fall into three general categories: the magnitude of impact, the quality of both the impact site and the potential offset site and the mitigation method used to create the offset.

The relevance of nearly all commonly-considered factors seems to be determined by the method used to calculate impacts and offset potential. As discussed above, the common practice of using habitat area as a proxy for BES may be expedient but is seldom effective, as it overlooks many factors that cause each target to vary within a habitat type. Most factors traditionally used to calculate biodiversity offset ratios are introduced to try to correct for these oversights, and establish more accurate impact estimates or more accurate equivalency between impact and offset sites. However, functional assessment approaches negate the need for these factors when they are directly incorporated into the assessment of impacts and offsets. For example, because area-based approaches do not estimate marginal impact, but rather assume full biodiversity loss, some ratio factors are used to adjust the actual impact based on the type of development (e.g. a higher ratio for a more damaging kind of development). If impact assessment accounts for the marginal change through a functional assessment, little adjustment would be needed via an offset ratio. Similarly, habitats of the same type, but farther away from the impact site may be assigned a higher offset ratio because of the chance that the habitat is too dissimilar to support the biodiversity lost. A functional assessment that accounted for the actual differences in habitat would not need to rely on a ratio to reflect this possibility. When such comprehensive functional assessments are not possible, ratio factors can be used to correct any shortfalls.

Some factors are relevant for ratio calculation even if a comprehensive functional assessment method is used, as they are difficult to account for. For example, all assessment methods are associated with some uncertainty, and best practice would be to increase the offset ratio in cases where it is known that datasets or models are more uncertain (Bull et al., 2013). As another example, it is possible to capture the risk of losing offset benefits due to natural disturbances or stochastic events in functional assessments, but likely challenging due to model and capacity limitations. This, and other similar factors may need to be accounted for in a ratio even when functional assessments are done.

For ES, the inclusion of some factors in ratio calculation will depend on whether the impact and offset assessments include value, or stop at service delivery. For example, all impact assessments have some assumed timeframe into the future over which the impact of the development is being captured. For ES impacts, the longer into the future the impact will persist before the offset fully restores the benefit to society, the more value is lost. Economists generally agree that people value a benefit more if they are able to experience it today than if they were to experience it in the future. To reflect this preference economists commonly use discount rates to reflect the difference in value attributed to benefits received today versus in the future. Ecosystem service impact and offset assessments that do monetary valuation of services and include a net present value estimate account for this difference in value preference, while impact assessments that do not use this method will miss it. Put another way, if ES impacts and offsets are not compared in value terms and a discount rate is therefore not used, then the assessment implicitly applies a discount rate of zero meaning that stakeholders do not have any preference for the timeframe of impacts and offsets that occur today relative to in the future. Many standard economic valuation methods include discounting, and some tools for compensation calculation do as well (e.g. Habitat Equivalency Analysis (HEA) and the Resource Equivalency Analysis (REA) (Snyder and Desvousges, 2013)). When such methods or tools are used, additional factors are not needed to adjust ratios for this concern.

Non-value based assessments could potentially approximate time rate preferences through mitigation ratios (as is done in some biodiversity assessments, e.g. Denne and Bond-Smith, 2012), though this would need to be explicitly discussed among stakeholders to ensure clarity on the purpose of this approach and context-specific preferences. The need for time discounting has also been recognized for biodiversity impacts given that the losses to species and habitats are often more certain than are the offset benefits in the future (e.g. given restoration time lags and uncertainty of success) (Overton et al., 2013).

Overall, there is no conceptual reason that the potential set of factors used in ratio determination for BES should vary dramatically from site to site. A consistent approach can be developed, and could start by

Table 1
Biodiversity mitigation ratio factors and their applicability for ecosystem services offsets.
Modified with permission from Mandle et al. 2013.

				Relevance to biodiversity		Relevance to ecosystem services			
Concept	Factor	Example metric	Condition associated with a higher offset	Area- or habitat-based	Functional assessment	Area- or habitat-based	Production function, delivery	Production function, value	Ecosystem service considerations
	Type of development	Mine, road, agriculture	More damaging type of development	Yes	No	Yes	No	No	
	Impact site condition	Wetland function at site prior to impact, value of service x at site prior to impact	Higher function	Yes	No	Yes	No	No	
	Vulnerability	Target loss rate(s) from other drivers, sensitivity of target(s) to type of development	More vulnerable targets	Yes	No	Yes	No	No	
Magnitude of impact	Duration of impact	Time frame of proposed development	Longer development term	Yes	No	Yes	No	No	
	Rarity of target(s)	Number of rare species at impact site	More rare species or habitats	Yes	No	Yes	Yes	No	Rarity is conceptually equivalent to service scarcity, where a service that is more scarce is more valuable. Valuation impact estimates would capture any stronger social preference for a more scarce service.
	Intergenerational equity	Strength of desire to serve future generations	Stronger desire	Yes	Yes	Yes	Yes	No	Monetary impact assessments that included a future time frame of impact and a discount rate would directly capture this.
	Uncertainty of impact estimate	Variance in impact estimates	Higher uncertainty	Yes	Yes	Yes	Yes	Yes	Most ecosystem service impact assessments are likely to have associated uncertainty that should be considered when setting an offset requirement.
	Similarity of site conditions (in-kind vs. out-of-kind)	Habitat type, level of function	Poorer match between sites	Yes	No	Yes	No	No	
	Distance from impact site	Distance, within impact watershed	Farther from impact site	Yes	Yes	Yes	No	No	Production-function approaches generally link services to beneficiaries in space, so would capture important serviceshed context. Area- or habitat-based assessments likely would not.
Quality of offset	Leakage	Probability of development shifting to another area	Higher probability of leakage	Yes	Yes	Yes	Yes	Yes	This is not currently applied to biodiversity, but is relevant. Most assessments do not capture probability of leakage.
	Risk of natural disturbance	Frequency of fire, freqeuncy of floods	Higher frequency of disturbance	Yes	Maybe	Yes	Maybe	Maybe	Even functional assessments of biodiversity and ecosystem services may not include natural disturbances in impact and offset estimates.
	Additionality	Probability of development, deforestation rate	Lower risk of development	Yes	Maybe	Yes	Maybe	Maybe	Functional assessments of offsets may not include future scenarios of cumulative impacts or losses due to other drivers of change. If these are captured in the offset estimate, then they do not need to be included in a ratio.
	Risk of failure	Historical success rate of method	Lower rate of success	Yes	No	Yes	No	No	
Mitigation method	Time lag of recovery	Time period to recover target(s) to desired level	Longer recovery time	Yes	No	Yes	No	No	
	Perpetuity	Historical average duration of method success	Shorter period of outcome persistence	Yes	No	Yes	No	No	

choosing factors for ratio calculation based on what is not captured in impact and offset assessments (e.g. Table 1). We have provided a starting point for such an approach, but further work is needed to develop a method for combining relevant factors into defensible equations that can be used to assign ratios in a rigorous and replicable approach.

3. Conclusions

Many calls have been made for the inclusion of ecosystem services in the mitigation hierarchy (Boyd and Wainger, 2003; Geneletti, 2011; Baker et al., 2013), yielding policy requirements in some arenas (IFC, 2012; Villarroya et al., 2014). We propose a comprehensive framework for advancing both biodiversity and ecosystem service mitigation. Despite challenges with implementing biodiversity mitigation, work to date forms a useful starting point for ES mitigation. Within the unified framework we present, we offer a means for choosing representative ES targets alongside biodiversity targets, define and identify servicesheds as a useful spatial unit for ES impact and offset assessment, and initiate a means for consistent calculation of BES mitigation ratios. We emphasize the need to move away from area- and habitat-based assessment methods for BES and towards functional assessments at landscape or seascape scales that more accurately and comprehensively reflect cumulative impacts and variation in environmental quality, social needs and value preferences.

Advancing this framework into practice will face several remaining scientific and political challenges. On the science side, we still know little about how to define appropriate BES thresholds that can help identify critical areas to avoid development (Huggett, 2005; Chaplin-Kramer et al., 2015). Worth special attention is the challenge in setting ES thresholds of how we weigh and capture potential future ES needs that will evolve with human population trajectories and shifting social preferences. Capacity, data and models still pose significant limitations for functional assessments of BES. Serious efforts must be placed on advancing these limitations as quickly as possible, as emerging data continue to show the dramatic shortfalls of the commonly used area- and habitat-based assessment approaches. Finally, existing data and models can and should be used now to further explore the spatial prioritization of offsets to clarify under which ecological and social contexts different prioritization approaches (e.g. biodiversity first, ES first, joint prioritization) will give the most efficient mitigation outcomes.

Clearly, there is still a need to crosswalk our general framework with the specific requirements and needs of different kinds of assessment (strategic environmental assessment (SEA), environmental impact assessment (EIA), and social impact assessments (SIA)) and with different governments' specific regulatory guidelines. Different types of assessments will require tailoring of the general considerations described here, as will most context-specific applications of BES assessment (Polasky et al., 2015). For example, SEAs are decision-support processes that integrate environmental and sustainability considerations at strategic, policy and programmatic levels whereas EIAs and SIAs do so at project-levels. As such, their temporal and spatial scopes will vary and in turn influence different relevant BES targets. For example, the consideration of ES aspects like water supply, coastal protection, and climate regulation may be better addressed with strategic-scale BES assessments if they do not deliver tangible benefits to current populations but are expected to increase in importance to society over the long-term.

The actors responsible for making decisions throughout the application of the framework, such as who determines a mitigation ratio, and how to align spatial and temporal scales considered in BES offsets, will also vary based on existing jurisdictions, requiring additional effort to determine how this general framework can best be applied under different regulatory contexts. Additionally, much legislation lacks a clear recognition and strong requirement for the establishment of avoidance areas before specific development projects are considered—a necessary requirement for many of our recommendations to come into play. Further, many countries stop their legal requirements at the impact assessment stage, failing to require compensatory mitigation for allowed impacts. Similarly missing are government regulations and capacity to move ES mitigation requirements from law into practice.

Despite the remaining challenges, impact assessment and mitigation processes provide one of the best opportunities to incorporate BES information into land use decisions through widely adopted environmental regulations. The establishment of ES concepts and approaches has brought renewed attention and opportunity to the improvement of impact assessment methods. The integrated framework we present builds on the experience of biodiversity mitigation and address the unique opportunities and challenges presented by ES mitigation. By conducting integrated BES impact assessments and mitigation actions, we will have a more robust opportunity to meet the joint environmental and social intents of most laws that call for them.

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