Title: The Risks and Opportunities of Translating Terrestrial Biodiversity Offsets to the Marine Realm

Authors: Nicole Shumway[1], James E.M. Watson[1, 2], Megan Saunders[1] and Martine Maron[1]

[1] School of Earth and Environmental Sciences & Centre for Biodiversity and Conservation Science, The University of Queensland, Brisbane, QLD 4072, Australia

[2] Wildlife Conservation Society, Global Conservation Program, Bronx, NY, USA 10406

Email: Nicole Shumway: n.shumway@uq.edu.au; James E.M. Watson: jwatson@wcs.org; Megan Saunders: m.saunders1@uq.edu.au; Martine Maron: m.maron@uq.edu.au

Key words: conservation policy, marine biodiversity offset, marine compensation, no net loss

Corresponding author: Nicole Shumway, School of Earth and Environmental Sciences, Room 327D, Steele Building (#3) The University of Queensland, Brisbane, QLD 4072, Australia

Author Biographical Information: *Nicole Shumway is a PhD Candidate, Martine Maron and James EM Watson are both Associate Professors and Megan Saunders is a Postdoctoral Research Fellow at the University of Queensland School of Earth and Environmental Sciences, and the Centre for Biodiversity and Conservation Science. JEMW is also Director of the Science and Research Initiative at the Wildlife Conservation Society.*

Abstract:

Biodiversity offset programs are expanding rapidly at a global scale in the marine realm, despite few data being available on their effectiveness. We reviewed the literature on biodiversity offsets to determine where marine offset policies occur, to what degree they are implemented and, using this information, identify the most important differences between marine and terrestrial systems that are likely to have implications for how offsetting is done. We found that 77 nations had compensatory policies that enabled the use of offsets in the marine environment. Two important differences between marine and terrestrial offsets emerged: 1) biophysical, such as greater connectivity, likelihood of restoration success and data paucity, and 2) social/governance, such as a lack of ownership and a greater probability of leakage. The lessons learned from the development of terrestrial offsets provide an opportunity to improve how they are applied to marine conservation and reduce net impacts on marine ecosystems.

Billions of people depend on marine and coastal systems for essential ecosystem services, including climate regulation and food resources (Costanza 1999). Yet this reliance has meant that marine ecosystems are increasingly being degraded by human activities (Halpern et al. 2008). As human populations grow, pressures on the marine environment will continue to increase, from direct impacts such as fishing and resource extraction, loss of habitat, and pollution, and indirectly through anthropogenic climate change (Halpern et al. 2007). These impacts vary in their intensity and spatial distribution across the seascape (Halpern et al. 2008), but marine industry and resource extraction is growing, especially in deep water and other remote and frontier areas that were previously inaccessible (Kark et al. 2015).

Biodiversity decline appears to be an inevitable consequence of this increased exploitation and pressure on the marine environment. Increasingly, industry is being expected not only to minimise impacts of their activities on the environment, but to act to counterbalance any residual impacts with the goal of achieving ‘no net loss’ (NNL) of biodiversity (BBOP, 2012b). The achievement of NNL is generally associated with the use of a mitigation hierarchy, where impacts are sequentially avoided, minimized, restored and lastly, offset. Biodiversity offsetting is a mechanism used to mitigate impacts from development on species and ecosystems, in theory allowing economic gains without the associated biodiversity loss (Maron et al. 2012). Biodiversity offsetting works by either restoring, rehabilitating or protecting comparable habitat (McKenney and Kiesecker 2010) in order to generate a biodiversity ‘gain’ equivalent to the loss from development.

Most offset policy development and research has focussed on terrestrial ecosystems and species. Marine biodiversity offsets remain an emerging mechanism for impact mitigation in most parts of the world, even where terrestrial offsets are prevalent. For example, in

Australia, terrestrial offsetting has occurred since 2000, with little marine offset policy development until recently (Dutson et al. 2015, Maron et al. 2016b). Even in areas where marine offsets have been used for decades (e.g., Canadian fish habitat mitigation), relevant data on the implementation and success have been difficult to obtain (Levrel et al. 2012).

There are major challenges associated with achieving NNL in biodiversity offsetting (Bull et al. 2013a). These include defining appropriate metrics to account for biodiversity losses and gains to achieve commensurate offset exchanges (Maron et al. 2016a), the determination of suitable frames of reference both currently and in the future (Maron et al. 2015), a gap between the theory and implementation of multipliers to account for uncertainty (Bull et al. 2016b), a lack of adherence to the mitigation hierarchy prior to offsetting, and a lack of monitoring of offsets that are already in place (Quigley and Harper 2006). In addition, Maron et al. (2016a) point out the ethical and social challenges involved in biodiversity offsetting, such as accurately reflecting societal values towards nature, and balancing the trade in nature with a moral obligations to protect biodiversity.

Many of the challenges of terrestrial offsetting are likely also to occur in marine and coastal environments. However, due to fundamental differences in both the ecology and governance of marine ecosystems, several of these challenges are likely to be more or less problematic, and completely new ones may emerge. It is important to identify these challenges before the widespread application of marine offsetting, in order to inform limits to offsetability in the marine realm, and to help improve the design of marine offset policy.

We first reviewed the incidence of compensatory policies in the marine environment to determine the scale of marine offsetting globally. Based on this literature review and the broader literature on biodiversity offsets, marine ecology and conservation, we compared the potentially important differences between the ecology and governance of terrestrial and marine ecosystems. In each case, we considered how those differences might influence the translation of offsetting to the marine environment in order to understand the unique challenges faced by the achievement of marine NNL, and outline the implications for marine offset design and implementation.

The occurrence of marine offsets globally

Data on 148 separate offset or compensatory policies were collected (see supplemental appendix 1 for methodology). These included national, regional and state policies, as well as marine offset policies for businesses (e.g. Royal Dutch Shell Pearl GTL Project, Qatar). A great deal of ambiguity exists around the terminology of ‘offset’ policies globally (Bull et al. 2016a), making the identification of a policy designed to achieve true ‘offsets’ with a NNL objective difficult to determine. In many countries the term offset is used synonymously with mitigation (e.g. United States wetland mitigation), or compensation (e.g. Canada fish habitat compensation). Therefore, we included policies with frameworks in place for offset development (e.g., environmental impact regulations) that included mention of offsets or compensation. Based on this broad definition, 77 countries had offsets occurring or compensatory policies in place or under development that involve offsets or some similar form of compensatory mechanism, and enable the use of offsetting or compensation in the marine environment (Fig. 1). This includes 22 member states of the European Union that have marine Natura 2000 sites, which require compensation for damage under the EU Birds and Habitats Directives. In addition, marine specific offset policies occur in 12 countries, though many of these countries also have multiple offset policies covering both national and sub-national jurisdictions.

Insert Figure 1 here.

Differences between marine and terrestrial systems

While terrestrial and marine offsets have the same theoretical basis, we identified important practical differences. These differences in ecological and biological processes can be categorized into two distinct categories, 1) biophysical, and 2) social/governance (those influenced by societal, legislative or management involvement). Several of these inherent differences between marine and terrestrial ecosystems have important implications for offset feasibility and effectiveness, and we discuss these in this section.

Biophysical challenge: strong connectivity between terrestrial and marine environments

Spatial and hydrological connectivity dominate the marine environment. The convection process of waves and currents in the ocean leads to more open systems with greater ‘flow’ (movement of water and organisms) than is common in terrestrial systems (Bos et al. 2014, Carr et al. 2003). Marine and coastal systems are highly connected to terrestrial systems and exchange everything from energy and materials to organisms. For example, the Great Barrier Reef contains 35 defined river basins (Furnas 2003), and just one (the Burdekin watershed), has been show to affect nearly 47,000km^2 of marine area, including 247 different reefs and 73 seagrass beds (Devlin et al. 2012). While generally most suspended solids and particulate nutrients are deposited within a few kilometres of a river mouth, inorganic nutrients can be conveyed over distances of 50 km or more, and high concentrations of dissolved nutrients can be measured up to 200 km from the river mouth (Devlin and Brodie 2005).

This connectivity can lead to circumstances where biodiversity in one system is threatened by impacts in another, such as the biological ‘dead zones’ seen in the Gulf of Mexico as a result of nutrient outflows from urbanization and upstream agriculture in the Mississippi river (Stoms et al. 2005), or the impact of deforestation on coral reefs in Fiji (Klein et al. 2012). As such, marine systems are highly influenced by human activities occurring at potentially great distances (Devlin et al. 2012). Defining and quantifying the impacts from a particular development, and developing and measuring offset benefits, could therefore be especially difficult due to the strong confounding influences from across the land-sea gradient. For example, it would be difficult to predict the impact of a single additional farm on the condition of a downstream coral reef in a catchment dominated by farming.

The achievement of NNL in the marine environment must therefore account for both direct and diffuse impacts, as well cumulative impacts from multiple stressors occurring in multiple locations. While these types of impacts may also affect terrestrial offsets, indirect impacts in the marine environment may be more common and more significant than direct impacts, an unusual occurrence in terrestrial environments. For diffuse impacts that occur as a result of decreased water quality (e.g. impacts to seagrass from increased sedimentation), offsets could occur on land or at sea and be theoretically feasible (Bell et al. 2014). For example, offsets could be a direct action through replanting of seagrass or indirect through increasing riparian vegetation cover to reduce sedimentation. However, much more research would be needed to be confident of achieving equivalence in such cases (i.e., that offsets on land for impacts at sea are measurable and equivalent).

Biophysical challenge: greater connectivity within the marine environment

Enigmatic impacts, such as indirect impacts outside of a development footprint, can be an important issue in terrestrial systems (Raiter et al. 2014); however, they may be particularly dominant in marine environments. Indirect impacts in the marine environment can have substantial effects in adjoining and even distant marine systems, and influence a greater number of environmental values. Repeated dredging of the sea floor for a port, for example, is likely to impact water quality far from the dredging 'footprint', leading to problems such as habitat degradation and alterations to feeding and predation behaviour at potentially great distances from the development (Raiter et al. 2014). For this reason, a more rigorous assessment of impacts is particularly important for the achievement of NNL in the marine environment, with a specific focus on indirect and offsite impacts to marine biodiversity. For impacts on biodiversity values that are locally important either socially or biologically, or that have a restricted range, offsets should continue to occur near the impact site (currently best practice) (Ives and Bekessy 2015). To account for wide-ranging impacts far from development footprints, there is increasing evidence that offsets could also be used to support more systematic conservation planning at a regional scale (Kujala et al. 2015).

Biophysical challenge: greater organism dispersal and migration

Marine systems are dominated by species with complex life histories, with most having at least one widely dispersive or migratory phase. This large-scale dispersal, often through the use of ocean currents, ultimately determines species distributions and is important for the maintenance of genetic diversity (Carr et al. 2003, Trakhtenbrot et al. 2005). For example, a large number of sedentary adult marine species produce planktonic young that disperse great distances (Carr et al. 2003): less than 1 km for some sessile species (corals, bryozoans, tunicates), but 20 km to hundreds or thousands of kilometres for other broadcast spawners such as fish, molluscs and crustaceans (Shanks et al. 2003). The dominance of large scale ecological connectivity observed within the marine environment means that geographical and political boundaries could be less relevant for marine offsetting. This may present an opportunity to introduce conservation interventions that counterbalance the impact by working either within the impacted region itself, or other areas important to the impacted species.

Many marine species have long-distance migrations that span numerous countries and jurisdictions, and are impacted by multiple threats throughout their range. While this is also true for terrestrial species (Bull J.W. et al. 2013b), it is more common in the marine environment (Carr et al. 2003). Conservation interventions can be particularly difficult to implement for ‘moving targets’ (Bull J.W. et al. 2013b, Runge Claire A et al. 2015). Bull J.W. et al. (2013b) suggests that applying offsets more dynamically anywhere impacts are occurring within a migratory species’ range or pathway, rather than just near the impact site, could provide better conservation outcomes. Offsets could then augment existing conservation interventions in areas of high threat, leading to more coordinated conservation networks at the landscape scale, and reducing the risk of offset failure by distributing that risk at a wider scale (Bull J.W. et al. 2015, BBOP, 2012a, Habib et al. 2013).

Because migratory species often depend on a series of interconnected sites, interventions at an unprotected link in the migratory pathway (Runge C. A. et al. 2014) could provide outcomes orders of magnitude better than if the same offset was placed near the impact site.

For instance, migratory shorebirds in the East Asian-Australasian Flyway (EAAF) have specific stop-over sites to rest and refuel during migration, but habitat loss at these sites has led to a significant bottleneck that threatens shorebird populations throughout the region (Murray et al. 2014). There is therefore an opportunity for an offset to generate a greater benefit for these migratory species by averting loss in areas where the threats are greatest or more tractable. There are also risks associated with allowing such spatial flexibility (i.e., implementation of offsets far from the impact site). First, equivalence between the biodiversity impacted and the biodiversity offset could become difficult to evaluate, obscuring the connection between biodiversity lost from the impact and gains accrued from the offset; offsets could also become difficult to track, monitor and manage, especially if the impact and the offset occur in different jurisdictions (Bull Joseph W et al. 2016a, Vaissière et al. 2014). Allowing offsets that are flexible in space could also lead to the loss of important ecosystem services and cultural values that may be socially unacceptable (Rogers and Burton 2016). Finally, it could exacerbate social inequalities, with developed countries continuing to develop, while encouraging offsets in less developed countries with high biodiversity (McDermott et al. 2013).

Biophysical challenge: ecological limitations to restoration in the marine environment

Active restoration or rehabilitation of already-degraded ecosystems is a key mechanism for achieving biodiversity gains to offset losses from development impacts (another is averting future loss, i.e., through protection) (Maron et al. 2012). Biodiversity offsetting in terrestrial environments relies heavily on restoration and is crucial to achieving offset objectives (Madsen et al. 2010, Maron et al. 2012). However, in most marine environments, restoration is considerably less effective and more expensive (Bayraktarov et al. 2016). One study reviewed seagrass restoration projects completed since 1990 in New South Wales (NSW), Australia, finding that most projects could not achieve their 2:1 offset-to-impact goal, and therefore current restoration cannot be relied upon to achieve the seagrass habitat compensation policy for the state (Ganassin and Gibbs 2008).

A review of marine coastal restoration worldwide found that success was highly variable depending on the ecosystem and the project location (coral – 64.5%, mangrove – 51.3%, seagrass – 38%, saltmarsh – 64.8%) (Bayraktarov et al. 2016). Rates of success in this study were item based (e.g. the number of seedlings surviving) and most were assessed in the short term (1-5 years), while success of offsets through restoration is more likely related to the success of the restoration project overall and the likelihood that it achieves its NNL objective in the long-term. In addition, the average cost of marine restoration for all systems was 1.6 million USD/ha (Bayraktarov et al. 2016). Cost increases and feasibility decreases as you move from shallow to deeper water, with one theoretical study suggesting deep sea restoration could be feasible, but would be orders of magnitude more expensive (Van Dover et al. 2014).

The feasibility of restoration in the marine environment relates directly to the offsetability of project impacts. There is significant uncertainty that restoration based offsets can be relied upon to achieve NNL objectives or provide genuine gains for marine systems. Therefore, techniques for rehabilitating marine environments and the science underpinning marine restoration need further development before marine restoration can be a widely used offsetting mechanism. Regulators should ensure that multipliers are used appropriately to account for rates of success and risk of catastrophic loss from unavoidable natural events (*e.g.*

cyclones, floods) (Bull J.W. et al. 2016b), and also ensure offsets are not just implemented but also evaluated in the long term to ensure a NNL outcome is achieved.

Biophysical challenge: lack of data and poor resolution of available data

A major challenge in implementing offsets and achieving NNL in marine environments is the paucity of data compared to terrestrial environments (UNEP-WCMC 2015). A recent study mapping global critical habitat in the marine environment found that there is a substantial lack of data, limiting how accurately marine biodiversity values can currently be represented (Martin et al. 2015). In addition, though deep pelagic marine systems are one of the largest habitats on earth, they are immensely under-represented in global data (Webb et al. 2010).

There is also a lack of finely resolved data in marine environments. In many terrestrial environments, ecosystems have been delineated at quite fine resolutions, based on community composition (Queensland Herbarium 2016) or change in extent (Hansen et al. 2013). For example, in the state of Queensland, Australia, terrestrial regional-ecosystem mapping is available to finely delineate vegetation communities (1,540 in the state and 172 in southeast Queensland region) based on geology, soil and bioregion (*e.g.* regional ecosystem 12.5.6 is a Eucalyptus-dominated (several species listed), open forest on deep red soil found in bioregion 12 (Southeast Queensland), land-zone 5 (old loamy sandy plains), with several sub-regional ecosystems listed based on species composition (Queensland Herbarium 2016). Conversely, though the Great Barrier Reef is one of the best monitored and managed reef systems in the world, comprehensive mapping of coral communities does not yet exist. Though coastal and shallow systems are relatively well-studied, finely-resolved communities of species have generally not been mapped. Metrics for monitoring of coral reefs focus on percent coral cover, but should also include factors like species assemblages and diversity (Bellwood et al. 2004). In terrestrial environments, biodiversity is often accounted for through the use of composite metrics, such as ‘habitat hectares’, which assess and scores habitat quality based on multiple attributes benchmarked against similar undisturbed habitat (McKenney and Kiesecker 2010). This paucity of data in the marine environment makes it difficult to explicitly account for impacts to ecological communities or to changes in species composition or richness, and makes like-for-like exchanges of biodiversity difficult to achieve in marine offsetting.

Change in biodiversity is difficult to estimate in any system as it complex and multidimensional, but especially in marine systems where there are few underlying data. To account for impacts to coral reef habitat, for example from dredging, an offset would either have to quantify change in reef condition using a combination of assessment metrics (i.e. composite metrics) or use a surrogate metric as a proxy for harder to measure values. Composite metrics, such as habitat equivalence analysis (HEA) are often used in natural resource damage assessments, such as ship groundings on coral reefs (Dunford et al. 2004). However the successful application of a composite metric relies heavily on extensive supporting research, which is often limited in the marine environment (Viehman et al. 2009). In addition, these types of composite metrics have not yet been used for diffuse impacts or for marine offsetting. Surrogate metrics such as change in water quality, are much easier to measure than composite metrics, but are fundamentally less precise in accounting for damage. For the previous example of impacts to coral from dredging, the end-point biodiversity damage (the reef) cannot be directly offset, so an intermediate factor (water quality) is used. Water quality is then the factor that will mediate the impact to reef condition. The use of a surrogate to account for biodiversity means that both the measuring of the impact and the impact alleviation are one step removed from the actual biodiversity value. The use of better or multiple surrogates may lead to a more representative or equivalent estimation of a biodiversity value (Quétier and Lavorel 2011), but the use of surrogates will always be an imprecise way to measure changes in biodiversity. Not only are impacts difficult to define in data poor systems but offset outcomes are equally difficult to evaluate, leading to situations where NNL is theoretically possible but challenging to measure with any certainty.

Social and governance challenge: lack of private ownership

Unlike terrestrial environments, ownership and legal protections in the marine environment are limited. In terrestrial offsets, proponents can buy and protect land, or pay another landowner to manage threats and perform restoration activities on their land and ensure ongoing protection (BBOP, 2012a). Ownership in marine environments is much less common, so in marine systems this type of offsetting is unlikely to occur. Therefore, the options for how offsets can be accomplished in the marine environment are different. In part, the lack of private ownership may improve the potential effectiveness of offsets, as only one entity (usually a government body) can regulate activities, making monitoring of compliance easier. Lack of management could be more challenging in areas of indistinct ownership, and while a government can designate areas for offset implementation, sustained legal protection is difficult to maintain without an ongoing, high-level of public support for the initiative (Dutson et al. 2015). For example, while marine parks and areas of marine tender can be designated, they can also be quickly downgraded and degazetted if industry interests object (Mascia and Pailler 2011).

Nations only manage marine systems and resources within 200 nautical miles of the cost, inside the country's Exclusive Economic Zone (EEZ). Outside of this area there are limited legal protections. These 'high seas' cover almost half of the earth's surface and as resources are exhausted in more accessible regions, increasing technological advances are leading to a surge in exploitation of the deep sea for fishing, minerals exploration and marine energy production (Kark et al. 2015). How might offsetting of impacts work in such a context?

Governance in the high seas is complex and based on the United Nations Convention on the Law of the Sea (UNCLOS), which allows for all States to exploit resources therein, but also an obligation to protect the marine environment (United Nations General Assembly, 2012). UNCLOS relegates specific activities to sector-based management by different organisations or conventions, leaving policy in the high seas fragmented, imbalanced to competing stakeholder interests, and lacking comprehensive management both spatially and by sector (Gjerde and Rulska-Domino 2012, UNEP-WCMC 2015). For example, in the high seas, shipping and its impacts are managed by the International Maritime Association (IMO), deep seabed mining is regulated by the International Seabed Authority (ISA), and fishing activities are managed through state run Regional Fisheries Management Organisations (RMFO) (Gjerde and Rulska-Domino 2012). Though a U.N. resolution to develop a marine biodiversity strategy for the high seas is ongoing, expanding resource exploitation outside of national jurisdictions will magnify impacts to marine biodiversity and become increasingly important in the achievement of marine NNL (UNEP-WCMC 2015). Given the lack of legislative control in areas of the high seas, there is limited potential to ensure the continued monitoring and management of offsets over the long-term. Because of the lack on comprehensive management of the global commons, it is unclear who would regulate offsets in the open ocean.

Social and governance challenge: greater likelihood of leakage in marine resource exploitation

Leakage, or the displacement or shifting of a damaging activity rather than its complete removal, is a known problem in protected areas and carbon offsetting but has rarely been quantified explicitly in biodiversity offset initiatives (Moilanen and Laitila 2016, Virah-Sawmy et al. 2014). Leakage in the ocean could be a more prominent issue than on land, linked to both the connectivity of marine environments, and to the lack of ownership. For an extractive industry like mining where the resource is fixed in one location, leakage issues will likely be the same at sea as it is on land. However, other resources in the ocean, such as fish populations, move and migrate unhindered by ownership boundaries, and exploitation is likely to follow that movement. For instance, most farmers who stop farming part of their land and revegetate that portion as a terrestrial offset, are unlikely to then buy, clear and farm additional land to make up for that loss. In the marine environment, the establishment of a no-take fishing zone as an offset will remove the threat of fishing from that single location, but may not reduce overall fish catch, merely shifting the pressure to elsewhere in the region. This effect has been documented in the effort redistribution of fishing fleets concentrated around the boundaries of no-take zones, where spill-over from the protected areas are likely to be greater (Gell and Roberts 2003).

Due to the lack of ownership of offsets in the marine environment and the potential for offsets to be lost through time, averted loss offsets are likely to make up a substantial portion of offset actions in marine areas. However, the type of activies being offset through averted loss, are likely to be the type of activities easily moved elsewhere. For example, vessel traffic removed from a particular area will simply shift the potential for vessel strikes to elsewhere in the region. Offset planning may therefore need to involve phasing out that component of industry where exploitation is likely to continue to occur as a result of leakage. For example, buying back active fishing licenses rather than establishing no-take zones would reduce the likelihood of leakage by ensuring a reduction in the overall total allowable fishing catch. If more permanent solutions like industry 'phase-outs' are unlikely or unable to occur, the potential for leakage needs to be factored into the additionality of the offset, with initiatives such as REDD (Reducing Emissions from Deforestation and Forest Degradation) providing a good framework for this type of systematic evaluation (Moilanen and Laitila 2016, Virah-Sawmy et al. 2014). Preventing leakage in marine offsets is key, and should be done by incorporating more robust solutions (i.e., permanent removal of threatening processes) to development impacts whenever possible.

Implications and the ways forward

Marine biodiversity offsets are occurring across Earth, with policies enabling the use of marine offsets nearly as developed as in their terrestrial counterparts. Yet there are almost no data on their effectiveness. We argue that both biophysical and governance differences between marine and terrestrial environments will create significant challenges for translating offsets to the marine environment. We have also identified potential opportunities afforded by the nature of marine systems. Nevertheless, significant gaps remain in the scientific foundations for marine offsetting.

Many of the theoretical and practical challenges faced in biodiversity offsets are shared by both marine and terrestrial systems, but there are important practical considerations that will present particular challenges to offsetting in the marine environment. For example, the lack of spatially-explicit data about marine systems may result in impacts that are difficult to quantify and equally difficult to offset effectively. There is also the possibility that other issues may become more important when attempting to achieve NNL in the marine environment, such as the scale of cumulative impacts or the influence of threshold dynamics and ‘tipping points’ in many marine systems (UNEP-WCMC 2015).

Given the complexity and inter-connectedness of marine systems, building spatial flexibility into marine offset actions has the potential to result in better outcomes for species and a greater likelihood of achieving NNL. This should be approached with consideration of the system-wide dynamics, rather than focused narrowly near the impact site. Similarly, many impacts in marine systems are a result of terrestrial based activities, and the integration of both land and sea mitigation activities into offsets could mediate complex impacts more effectively. It is clear the science underpinning this land-sea connection needs to be explored more fully. In the terrestrial environment, the integration of offsets with conservation planning have been labelled ‘strategic’(Kujala et al. 2015, Sochi and Kiesecker 2016) and common sense tells us that it is best to implement offset actions where they will have the greatest chance of achieving NNL, especially in the marine environment where offset actions may be limited by lack of data, lack of ownership and poor restoration potential. The uncertainty of achieving NNL given the compounding biophysical and governance challenges of offsetting in the marine environment, demonstrates the need to focus on the entire mitigation hierarchy more fully. The precautionary principle is key, and more explicit emphasis is needed on the avoidance step of the mitigation hierarchy in achieving marine NNL (Phalan et al. 2017).

Lack of outcome reporting and transparency is a significant barrier to improving offset outcomes and achieving NNL of biodiversity. Canada has had mitigation strategies in place to offset impacts to marine and aquatic fish habitat since 1986, but a review of this policy found that most offset projects (86%) could not be evaluated due to a lack of monitoring data (Harper and Quigley 2005, Quigley and Harper 2006). While Environmental Impact Assessment (EIA) or similar documents in many countries are available for review, most offset management or outcome reports are not publically available and are therefore unable to be evaluated. Because of these problems, ex-post evaluation of marine offsets - the evaluation of the impact and the degree to which is has been mitigated by offsets - is currently challenging.

There is no doubt that the concept of no net loss has the potential to generate better conservation outcomes as coastal and marine development proliferates. But without emphasis on robust, scientific evaluation of offset outcomes and better innovation for restoration and other conservation interventions, biodiversity offsets in the marine environment work only in theory.

Acknowledgements: We would like to thank H. Possingham, J. Bull, C. Kuempel and P. Addison for valuable discussions on this topic. NS is supported by an Australian Government Research Training Program (RTP) Scholarship, and a University of Queensland Centennial Scholarship. MM is supported by Australian Research Council Future Fellowship grant FT140100516.

References

Bayraktarov E, Saunders MI, Abdullah S, Mills M, Beher J, Possingham HP, Mumby PJ, Lovelock CE. 2016. The cost and feasibility of marine coastal restoration. Ecol Appl 26:1055-1074.

Bell J, Saunders M, Lovelock CE, Possingham H. 2014. Legal frameworks for unique ecosystems – how can the EPBC Act offsets policy address the impact of development on seagrass.

Bellwood DR, Hughes TP, Folke C, Nystrom M. 2004. Confronting the coral reef crisis. Nature 429:827-833.

Bos M, Pressey RL, Stoeckl N. 2014. Effective marine offsets for the Great Barrier Reef World Heritage Area. Environmental Science & Policy 42:1-15.

Bull JW, Gordon A, Watson JE, Maron M. 2016a. Seeking convergence on the key concepts in 'no net loss' policy. Journal of Applied Ecology 53:1686-1693.

Bull JW, Hardy MJ, Moilanen A, Gordon A. 2015. Categories of flexibility in biodiversity offsetting, and their implications for conservation. Biological Conservation 192:522-532.

Bull JW, Lloyd S, Strange N. 2016b. Implementation gap between the theory and practice of biodiversity offset multipliers. Conservation Letters.

Bull JW, Suttle KB, Gordon A, Singh NJ, Milner-Gulland EJ. 2013a. Biodiversity offsets in theory and practice. Oryx 47:369-380.

Bull JW, Suttle KB, Singh NJ, Milner-Gulland EJ. 2013b. Conservation when nothing stands still: moving targets and biodiversity offsets. Frontiers in Ecology and the Environment 11:203-210.

Business and Biodiversity Offsets Programme (BBOP). 2012a. Guidance Notes to the Standard on Biodiversity Offsets. Washinton, D.C.: BBOP. Report no.

---. 2012b. Standard on Biodiversity Offsets. Washington, D.C. Report no.

Carr MH, Neigel JE, Estes JA, Andelman S, Warner RR, Largier JL. 2003. Comparing Marine and Terrestrial Ecosystems: Implications for the Design of Coastal Marine Reserves. Ecological Applications 13:S90-S107.

Costanza R. 1999. The ecological, economic, and social importance of the oceans. Ecological Economics 31:199-213.

Devlin MJ, Brodie J. 2005. Terrestrial discharge into the Great Barrier Reef Lagoon: nutrient behavior in coastal waters. Mar Pollut Bull 51:9-22.

Devlin MJ, McKinna LW, Alvarez-Romero JG, Petus C. 2012. Mapping the pollutants in surface riverine flood plume waters in the Great Barrier Reef, Australia. Mar Pollut Bull 65:224-235.

Dunford RW, Ginn TC, Desvousges WH. 2004. The use of habitat equivalency analysis in natural resource damage assessments. Ecological Economics 48:49-70.

Dutson G, Bennun L, Maron M, Brodie J, Bos M, Waterhouse J. 2015. Determination of suitable financial contributions as offsets within the Reef Trust. Report no.

Furnas M. 2003. Catchments and corals: terrestrial runoff to the Great Barrier Reef. Australian Institute of Marine Science & CRC Reef Research Centre.

Ganassin C, Gibbs PJ. 2008. A Review of Seagrass Planting as a Means of Habitat Compensation Following Loss of Seagrass Meadow NSW Department of Primary Industries. Report no.

Gell FR, Roberts CM. 2003. Benefits beyond boundaries: the fishery effects of marine reserves. Trends Ecol Evol 18:448-455.

Gjerde KM, Rulska-Domino A. 2012. Marine protected areas beyond national jurisdiction: some practical perspectives for moving ahead. The International Journal of Marine and Coastal Law 27:351-373.

Habib TJ, Farr DR, Schneider RR, Boutin S. 2013. Economic and ecological outcomes of flexible biodiversity offset systems. Conserv Biol 27:1313-1323.

Halpern BS, Selkoe KA, Micheli F, Kappel CV. 2007. Evaluating and Ranking the Vulnerability of Global Marine Ecosystems to Anthropogenic Threats. Conservation Biology 21:1301-1315.

Halpern BS, et al. 2008. A Global Map of Human Impact on Marine Ecosystems. Science 319:948.

Hansen MC, Potapov PV, Moore R, Hancher M, Turubanova S, Tyukavina A, Thau D, Stehman S, Goetz S, Loveland T. 2013. High-resolution global maps of 21st-century forest cover change. Science 342:850-853.

Harper DJ, Quigley JT. 2005. No net loss of fish habitat: a review and analysis of habitat compensation in Canada. Environ Manage 36:343-355.

Ives CD, Bekessy SA. 2015. The ethics of offsetting nature. Frontiers in Ecology and the Environment 13:568-573.

Kark S, Brokovich E, Mazor T, Levin N. 2015. Emerging conservation challenges and prospects in an era of offshore hydrocarbon exploration and exploitation. Conserv Biol 29:1573-1585.

Klein CJ, Jupiter S, Selig ER, Watts ME, Halpern BS, Kamal M, Roelfsema C, Possingham H. 2012. Forest conservatin delivers highly variable coral reef conservation outcomes. Ecological Applications 22:1246-1256.

Kujala H, Whitehead AL, Morris WK, Wintle BA. 2015. Towards strategic offsetting of biodiversity loss using spatial prioritization concepts and tools: A case study onmining impacts in Australia. Biological Conservation.

Levrel H, Pioch S, Spieler R. 2012. Compensatory mitigation in marine ecosystems: Which indicators for assessing the “no net loss” goal of ecosystem services and ecological functions? Marine Policy 36:1202-1210.

Madsen B, Carroll N, Moore Brands K. 2010. State of Biodiversity Markets Report- Offset and Compensation Programs Worldwide. Report no.

Maron M, Bull JW, Evans MC, Gordon A. 2015. Locking in loss: Baselines of decline in Australian biodiversity offset policies. Biological Conservation 192:504-512.

Maron M, Hobbs RJ, Moilanen A, Matthews JW, Christie K, Gardner TA, Keith DA, Lindenmayer DB, McAlpine CA. 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. Biological Conservation 155:141-148.

Maron M, et al. 2016a. Taming a Wicked Problem: Resolving Controversies in Biodiversity Offsetting. BioScience 66:489-498.

Maron M, Walsh M, Shumway N, Brodie J. 2016b. Reef Trust Offsets Calculator: A prototype calculation approach for determining financial liability for marine biodiversity offsets voluntarily delivered through the Australian Government Department of the Environment (Reef Trust). Cairns: Reef and Rainforest Research Centre Limited. Report no.

Martin CS, et al. 2015. A global map to aid the identification and screening of critical habitat for marine industries. Marine Policy 53:45-53.

Mascia MB, Pailler S. 2011. Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. Conservation Letters 4:9-20.

McDermott M, Mahanty S, Schreckenberg K. 2013. Examining equity: a multidimensional framework for assessing equity in payments for ecosystem services. Environmental Science & Policy 33:416-427.

McKenney BA, Kiesecker JM. 2010. Policy development for biodiversity offsets: a review of offset frameworks. Environ Manage 45:165-176.

Moilanen A, Laitila J. 2016. FORUM: Indirect leakage leads to a failure of avoided loss biodiversity offsetting. Journal of Applied Ecology 53:106-111.

Murray NJ, Clemens RS, Phinn SR, Possingham HP, Fuller RA. 2014. Tracking the rapid loss of tidal wetlands in the Yellow Sea. Frontiers in Ecology and the Environment 12:267-272.

Phalan B, Hayes G, Brooks S, Marsh D, Howard P, Costelloe B, Vira B, Kowalska A, Whitaker S. 2017. Avoiding impacts on biodiversity through strengthening the first stage of the mitigation hierarchy. Oryx:1-9.

Queensland Herbarium. 2016. Regional Ecosystem Description Database (REDD). Version 10.0 (December 2016) DSITI: Brisbane.

Quétier F, Lavorel S. 2011. Assessing ecological equivalence in biodiversity offset schemes: Key issues and solutions. Biological Conservation 144:2991-2999.

Quigley JT, Harper DJ. 2006. Effectiveness of Fish Habitat compensation in Canada in achieving no net loss. Environental Management 37:351-366.

Raiter KG, Possingham HP, Prober SM, Hobbs RJ. 2014. Under the radar: mitigating enigmatic ecological impacts. Trends Ecol Evol 29:635-644.

Rogers A, Burton M. 2016. Public preferences for the design of biodiversity offset policies in Australia. Crawley, Australia: University of Western Australia. Report no.

Runge CA, Martini TG, Possingham HP, Willis SG, Fuller RA. 2014. Conserving mobile species. Frontiers in Ecology and the Environment 12:395-402.

Runge CA, Watson JE, Butchart SH, Hanson JO, Possingham HP, Fuller RA. 2015. Protected areas and global conservation of migratory birds. Science 350:1255-1258.

Shanks AL, Grantham BA, Carr MH. 2003. Propagule Dispersal Distance and the Size and Spacing of Marine Reserves. Ecological Applications 13:S159-S169.

Sochi K, Kiesecker J. 2016. Optimizing regulatory requirements to aid in the implementation of compensatory mitigation. Journal of Applied Ecology 53:n/a-n/a.

Stoms DM, et al. 2005. Integrated coastal reserve planning: making the land-sea connection. Frontiers in Ecology and the Environment 3:429-436.

Trakhtenbrot A, Nathan R, Perry G, Richardson DM. 2005. The importance of long-distance dispersal in biodiversity conservation. Diversity and Distributions 11:173-181.

UNEP-WCMC. 2015. Marine No Net Loss: A feasbility assessment of implementing no net loss of biodiversity in the sea. Cambridge, UK. Report no.

Vaissière A-C, Levrel H, Pioch S, Carlier A. 2014. Biodiversity offsets for offshore wind farm projects: The current situation in Europe. Marine Policy 48:172-183.

Van Dover CL, et al. 2014. Ecological restoration in the deep sea: Desiderata. Marine Policy 44:98-106.

Viehman S, Thur SM, Piniak GA. 2009. Coral reef metrics and habitat equivalency analysis. Ocean & Coastal Management 52:181-188.

Virah-Sawmy M, Ebeling J, Taplin R. 2014. Mining and biodiversity offsets: A transparent and science-based approach to measure “no-net-loss”. J Environ Manage 143:61-70.

Webb TJ, Berghe EV, O'Dor R. 2010. Biodiversity's big wet secret: the global distribution of marine biological records reveals chronic under-exploration of the deep pelagic ocean. PLoS One 5:e10223.

Figure 1. Countries with marine offsets occurring or compensatory policies in place or under development that enable the use of offsetting in the marine environment, where a) ‘Established’ = countries with both a policy in place and offsets occurring (n=15); b) ‘Under Development’ = countries with a compensatory policy in place but no marine offset actions yet occurring (n=16); c) ‘Preliminary’ = countries with no compensatory policy in place, however policy discussions or development are occurring (n=15); and d) ‘Potential’ = countries with an enabling policy framework in place but no offset discussions yet occurring (n=18). Countries with hatching over shading represent EU member nations with marine Natura 2000 sites, which require compensation for damage under the EU Birds and Habitats Directives (n=22).