A guide to modelling priorities for management of land-based impacts on coastal ecosystems

Christopher J. Brown[1], Stacy D Jupiter[2], Simon Albert[3] , Ken RN Anthony[4], Richard J Hamilton[5, 6], Alexa Fredston-Hermann[7], Benjamin S. Halpern[7,8,9], Hsien-Yung Lin[10], Joseph Maina[11], Sangeeta Mangubhai[2], Peter J. Mumby[12], Hugh P. Possingham[5, 8, 13], Megan I. Saunders[14], Vivitskaia JD Tulloch[13,15], Amelia Wenger [16], Carissa J. Klein[16]

1. Australian Rivers Institute, Griffith University, Nathan, Queensland, 4111, Australia
2. Wildlife Conservation Society, Melanesia Program, 11 Ma'afu St, Suva, Fiji
3. School of Civil Engineering, The University of Queensland, Brisbane, 4072, Australia
4. Australian Institute of Marine Science, Townsville, Queensland, 4810, Australia
5. The Nature Conservancy, Asia Pacific Resource Centre, 48 Montague Road, QLD 4101, South Brisbane, Australia
6. ARC Centre of Excellence for Coral Reef Studies, James Cook University, QLD 4811, Townsville, Australia
7. Bren School of Environmental Science & Management, University of California Santa Barbara, Santa Barbara, California, USA.
8. Imperial College London, Silwood Park Campus, Buckhurst Road, Ascot SL57PY, UK
9. National Center for Ecological Analysis & Synthesis, Santa Barbara, CA 93101, USA
10. Quantitative Fisheries Center, Michigan State University, 480 Wilson Road , 13 Natural Resources Building, East Lansing, MI, USA
11. Department of Environmental Sciences, Macquarie University, Sydney
12. Marine Spatial Ecology Laboratory, School of Biological Sciences, The University of Queensland, St Lucia, 4072 Australia

13. Centre for Biodiversity and Conservation Science, School of Biological Sciences, University of Queensland, St. Lucia, 4072 Australia
14. School of Engineering, University of Queensland, St. Lucia, 4072 Australia
15. Marine Predator Research Group, Department of Biological Sciences, Macquarie University, NSW 2109, Australia
16. School of Earth and Environmental Sciences, The University of Queensland, Brisbane, 4072, Australia

Abstract

1. Pollution from land-based run-off threatens coastal ecosystems and the services they provide, detrimentally impacting the livelihoods of millions of coastal people. Planning for linkages among terrestrial, freshwater and marine ecosystems can help managers mitigate the impacts of land-use change on water quality and coastal ecosystem services.
2. Here we examine the approaches used for land-sea planning, with particular focus on the approaches currently used to estimate the impacts of land-use change on water quality and fisheries, although our findings could also be applied to other ecosystem services. The review encompasses: modelling of large scale drivers of land-use change; local activities that cause such change; run-off, dispersal and transformation of pollutants in the coastal ocean; ecological responses to pollutants; socio-economic responses to ecological change; and finally the design of a planning response.
3. We find there is a disconnect between the dynamical models that can be used to link land to sea processes and the simple tools that are typically used to inform planning. This disconnect may weaken the robustness of plans to manage dynamic processes.

Land-sea planning is highly interdisciplinary, making the development of effective plans a challenge for small teams of managers and decision-makers.

Synthesis and applications

We therefore propose some guiding principles for where and how dynamic land-sea connections can most effectively be built into planning tools. Tools that can capture pertinent processes are needed, but they must be simple enough to be implemented in regions with limited resources for collecting data, developing models and developing integrated land-sea plans.

Keywords

ridge-to-reef planning, conservation planning, fisheries, run-off, integrated coastal management

Introduction

Coastal ecosystem services support the livelihoods of millions of people globally, but are threatened by multiple human pressures (Halpern *et al.* 2009). The close proximity of human settlements to coastal ecosystems means they are often exposed to both intense fishing pressure and run-off of land-based pollutants, among many other human pressures. The management of fishing pressure is supported by mathematical models that provide quantitative advice for how regulations should respond to changes in fish population size (Hilborn & Walters 2013). The management responses to land-based pollutants are less clear, in part because there are no standard models for quantifying the effects of land-use change on coastal ecosystems. The discipline of land-sea planning attempts to provide quantitative guidance for management decisions, typically through spatially-explicit recommendations for management actions (Alvarez-Romero *et al.* 2011).

Management responses to the impacts of pollution run-off on fisheries are difficult to formulate because the cause of pollution on land is separated in time and space from its impacts on fisheries by a chain of processes (Alvarez-Romero *et al.* 2011). Tracing the impacts of pollutants on fisheries require: (1) quantifying water quality changes resulting from upstream activities (e.g., forest conversion, agriculture, changes in land-use practices, coastal development); (2) determining the dispersion and transformation pathways of point- and non-point sources of pollutant plumes in the ocean in light of coastal hydrodynamics; (3) quantifying the direct and indirect ecological effects of pollutants on fish under different exposure regimes; and (4) translating ecological impacts on fish populations into economic consequences for coastal ecosystem services (Figure 1). Additionally, the long-term prediction of pollutant impacts may often require considering large scale drivers of land-use change, like climate change or global market forces (Figure 1). Connecting this chain of processes is empirically challenging, and few studies have been able to link changes in catchments to changes in coastal ecosystem services (Alvarez-Romero *et al.* 2011). This knowledge gap hinders land-sea management, because plans cannot be made that account for the dynamics of land-sea connections.

In recent years there has been increasing demand for quantitative models to support evidence-based planning (Carroll *et al.* 2012). Quantitative models can aid planning because they allow prediction of the outcomes of ecological or economic responses to land-use change. Linking land-uses to fisheries requires an interdisciplinary approach, because models must cover land-use change, pollutant paths, physical and chemical oceanography, fisheries ecology, economics and social science. Traversing these different disciplines is a challenge for the small teams running on tight time schedules who may typically develop land-sea plans

(Brown *et al.* 2017a). One source of land-sea models is the science of integrated coastal zone management, which has contributed greatly to our understanding of social and institutional pressures (Christie *et al.* 2005). However, the integrated coastal zone management literature has mostly provided qualitative models (e.g. Stoms *et al.* 2005). Qualitative models can be used suggest precautionary management guidelines or rank environmental state on an ordinal scale, but should not be used in planning algorithms because their combination with other data sources can become arbitrary (Game, Kareiva & Possingham 2013).

Here we propose a way forward for developing integrated models that can inform land-sea plans for managing coastal ecosystem services. First we review the peer-reviewed literature for studies that have developed quantitative approaches to spatial planning that bridge the land-sea divide. We then review models for connecting land to sea processes (Figure 1) with a focus on how representation of process dynamics may aid in predicting outcomes and change the conclusions of a planning process. Throughout, we focus on the models and tools used in various disciplines and how they can be integrated. We take a model-centric approach because quantitative models can be integrated directly into planning tools and used, for instance, to estimate the cost of land-use change to fisheries (Knowler *et al.* 2003) or provide information about planning alternatives (Arkema *et al.* 2015). Finally, we make recommendations for how future planning processes can balance model complexity with the need to develop integrated land-sea management strategies on the time-scales required to make decisions about local actions and policy development.

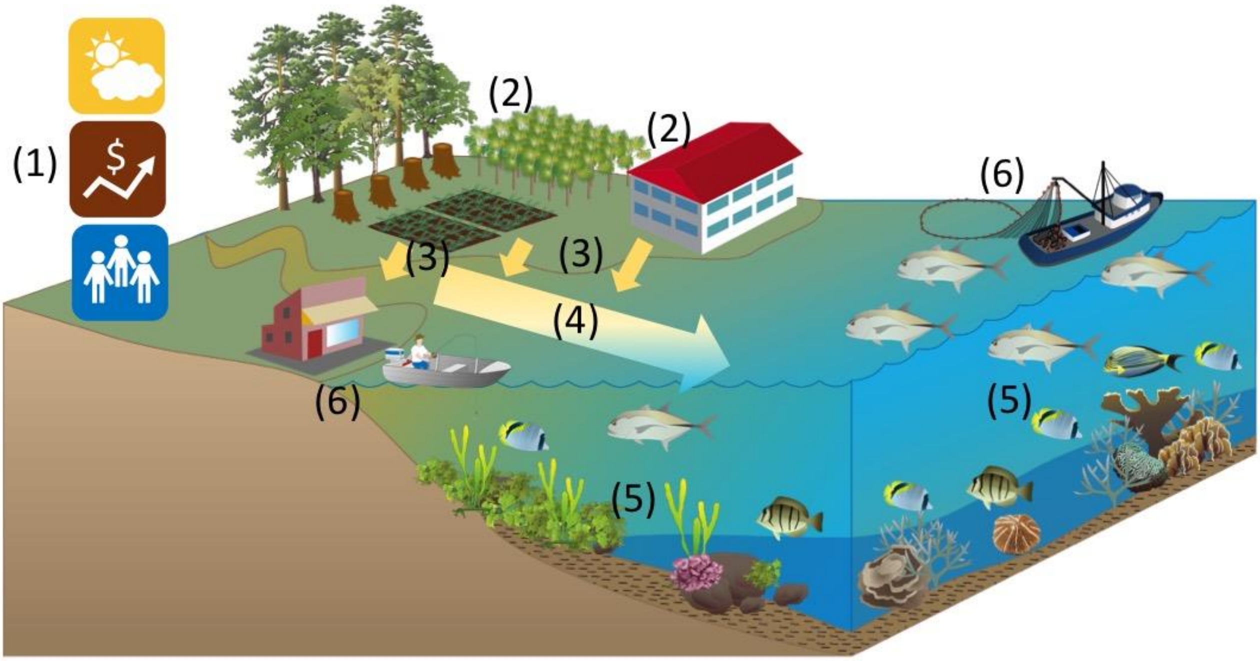

Figure 1 Conceptual overview of land-to-sea connections: (1) climate, economic and societal drivers of land-use change; (2) human activities that change pollutant run-off, including forestry, agriculture and urbanisation; (3) sediment and nutrient run-off from activities on land enter streams and eventually the ocean; (4) resulting changes in water quality as pollutants are dispersed and transformed in the ocean; (5) changes in marine ecosystems and fished populations, including interactions between predators, prey and between fished species and their habitats; and (6) impacts of ecological change on fisheries and social and economic responses to change in fisheries. Images: Tracy Saxby, Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/imagelibrary/)

Planning for land-use change

Incorporating models on land-use impacts to fisheries into a quantitative planning framework provides a transparent and repeatable approach to planning (Game, Kareiva & Possingham

2013). We searched the literature for peer-reviewed examples of spatial land-sea plans and found 14 examples (Table S1). Numerous planning approaches have been developed, but vary in terms of what models were used to inform the process (Table S1).

In general, existing planning approaches can be divided into two categories: threat-based and outcome-based (Giakoumi *et al.* 2015). A threat-based approach aims to reduce the amount of threat to marine ecosystem(s) and species (e.g. Klein *et al.* 2010). An outcome-based approach aims to maintain or improve the state (e.g. health) of marine ecosystem(s) or species through the reduction of threats (e.g. Klein *et al.* 2012). We advocate moving towards outcome-based approaches, because they are directly connected to the ultimate objectives of planning and avoid nominal variables which may have unclear and unquantified relationships with a planner's objectives. However, we acknowledge that tracking threat mitigation can be useful for indicating likely progress towards ultimate goals when data on outcomes are limited.

Most land-sea conservation planning has focused on threat-based approaches, probably because outcome-based approaches require more data and/or modelling of processes (Table S1). An implicit assumption with many threat-based approaches is that threats relate linearly to the desired outcome of interest (i.e., the more threat is reduced, the greater the likelihood that the marine ecosystem will transition to a desirable state). The assumption of a linear relationship between threat and outcome may be violated in many circumstances, for instance where ecological or economic tipping points drive non-linear change in fisheries (Selkoe *et al.* 2015). Thus, developing outcome-based approaches is a high priority for land-sea planning and several are under development.

An outcome-based approach was used to determine where the protection or restoration on land versus in the sea can deliver the greatest return on investment for improving the area of seagrass meadows (Saunders *et al.* (2017b). To assess the benefits of conservation actions to seagrass. aa model that related the threats of sediment run-off from on land and anchor-chain dragging on seagrass was used. A key difference with outcome-based approaches when compared to threat-based approaches is that the models connecting processes to condition allow one to assess how actions may have nonlinear benefits at larger scales as the amount of action increases. For instance, change in extent of seagrass meadows in over time likely has a non-linear relationship with sediment run-off (Saunders *et al.* 2017a). Therefore, outcome-based approaches may result in fewer actions than threat-based approaches, because outcome-based approaches will target areas and levels of action where conservation will have greater return on investment (Giakoumi *et al.* 2015).

The main barriers for the use of outcome-based approaches in land-sea planning are reliable models and data that can predict how the type and quantity of threats impact a marine ecosystem. Moving away from threat-based towards outcome-based approaches is an area of further research that is making substantial progress as the field develops (Saunders *et al.* 2017b). Thus, we now review models that could be used to address processes from land-use change to change in ocean ecosystems and the sustainability of fisheries.

Modelling the land-sea-fisheries process

We found several examples where quantitative models have been used to link land-use change to ocean conservation (Table 1). These cases differed from those in the above analysis in that they did not include solely spatial conservation plans (Table S1). Below we summarize these examples to illustrate the current state of integrated land-sea modelling, identify opportunities to leverage and expand these examples into new contexts, and highlight important research gaps.

Table 1 Examples of quantitative land-sea studies that have linked land-use change to coastal ecosystems and fisheries. Blue boxes indicate steps where a specific quantitative model was used. Empty boxes indicate that no quantitative model was used, though that step may have been considered qualitatively.

Example	Drivers of land-use change	Human activities that cause land-use change	Sediment and nutrient run-off	Dispersion of pollutants in the ocean	Ecosystem response	Socio-economic response	Planning	Reference(s)
Fiji coral reefs								(Brown *et al.* 2017a;Brown *et al.* 2017b)
Fiji coral reefs								(Klein *et al.* 2010;Klein *et al.* 2012)
Coastal Ecosystems, New Ireland Province, Papua New Guinea								(Tulloch *et al.* 2016)
Southern Chile, Loco Fishery								(Van Holt *et al.* 2012)
Gulf of California								(Álvarez-Romero *et al.*

marine ecosystems								2015)
Gulf of Mexico Shrimp fishery								(Huang & Smith 2011)
Hawaii run-off to coral reefs								(Oleson *et al.* 2017)
Moreton Bay seagrass								(Saunders *et al.* 2017b)
Guam coral reefs								(Weijerman, Fulton & Brainard 2016)
Western Philippines coral reefs								(Melbourne-Thomas *et al.* 2011)
Hawaii groundwater runoff to reefs								(Delevaux *et al.* 2018)

Drivers of land-use change

Drivers of land use change can be global or local, and can stem from changes in climate, ecological, economic or policy processes. Global markets are a significant driver of land conversion from native biomes such as forests, and wetlands, to commodity crops (Lambin & Meyfroidt 2011). For instance, demand for oil palm is driving widespread deforestation in

Indonesia, Malaysia and Papua New Guinea (Fitzherbert *et al.* 2008). Land-use change can also be driven by global shifts in governmental policy. Modelling the drivers of land-use change can be useful to devise scenarios for land-sea plans.

Direct incorporation of drivers into planning algorithms could be used to devise alternate plans that respond to possible future government policies, such as subsidies that promote conversion of land to biofuel crops, like oil palm (Castiblanco, Etter & Aide 2013). Notably, none of the examples we examined included quantitative models of large scale drivers (Table 1). Disregarding long-term changes in drivers when devising land-sea plans may mean that the management actions become ineffective if the large-scale drivers cause a change in the system's dynamics. A less technically challenging approach, that still acknowledges long-term change in drivers, is to develop multiple scenarios of change and use these to inform models for the direct impacts of land-use change on ocean ecosystems. For instance, models of global climate have been used to inform scenarios for population growth and land-use change in the Great Barrier Reef's catchments predict outcomes for coral reef cover to 2100 under different governmental policies (Bohensky *et al.* 2011).

Human activities that cause land-use change

The spatial and temporal patterns in the activities causing land-use change are important determinants of pollutant run-off. Models of land-use change are commonly used in integrated land-sea models (Table 1). For instance, logging on steep slopes will tend to produce much greater amounts of sediment in run-off than logging on shallow slopes. The type of land-use is also important, for instance, cropland and urban areas will have very different nutrient run-off rates, so determining which land-use a patch of forest is most likely to be converted to is important (Álvarez-Romero *et al.* 2015). Thus, determining the type of land-use change and when and where it will occur is critical to predict pollutant run-off.

When and where land-use change occurs depends on the interaction among large scale drivers of land-use change, like economic demand for an agricultural product, and local factors, including existing land-uses, value of land for different economic uses, accessibility for machines and people, and cultural and natural values. For instance, there can be large variability in the suitability of different geographies for oil palm plantations. Plantations are most profitable when sited on highly acidic mineral soils and near existing infrastructure for transporting and processing palm oil (Comte *et al.* 2012). These local factors will determine the siting of plantations within a region.

Models of land-use change often compare different development scenarios, where drivers of land-use change are used to define the scenarios (e.g. Bohensky *et al.* 2011). At a finer scale, the conversion of particular parcels of land is defined by local factors, such as land tenure systems or profitability of the land for a particular use. For instance, a predictive analysis of expansion of oil palm across Indonesia defined five development scenarios, which represented different government policies (the drivers), covering plausible future policies that prioritised oil palm production, food production, biodiversity conservation and protection of peat soil carbon stocks (Koh and Ghazoul 2011). Under each scenario, expansion of oil palm across over 500 000 spatial units was then modelled on the basis of predicted profitability for oil palm. Profitability was predicted using GIS layers of soil types and rainfall and verified against maps of existing oil palm plantations.

Mapping existing land-uses is critical to defining future development and for estimating the status-quo for pollutant run-off. Satellite data are commonly used to map existing land-uses (Brown *et al.* 2017a), however land-use maps from governmental repositories may also be used (e.g. Álvarez-Romero *et al.* 2015). Satellite data can provide coverage across large areas, however their accuracy is subject to many caveats. For instance, differences in atmospheric conditions, vegetation phenology, and soil moisture can bias the classification of land-uses, which in turn affect the predictions of erosion in hydrological simulations (McCallum et al. 2006). Further work is needed to explore incorporating uncertainty in land-use classification into models of pollutant run-off.

Predictive models of land-use change commonly rely on economic data, such as the value of land for production forestry and the cost of access (Álvarez-Romero *et al.* 2015). Ideally, regional models of land-use change would use the same decision process as employed by the agencies carrying out those land-use changes. However, information about the areas a certain agency, such as a logging company, has prioritised for harvest is commercial in confidence and consequently not commonly available to researchers. Sometimes this information can be obtained by inviting industry to participate as stakeholders in the planning approach. Another approach is to derive an agency's priorities independently, based on past land-use change. For example, soil type, aspect and proximity to processing plants have been used to identify likely locations of oil palm plantations (Tulloch *et al.* 2016). The likely location of plantations was then used to assess the impacts of plantations on run-off of sediments to reefs.

Sediment and nutrient run-off from land

Hydrological and geochemical modeling has a long history but only in the past decade have these models been applied in land-sea planning (Table 1). The complexity of processes linking pollutant run-off to land-uses has resulted in divergent approaches to modeling run-off for land-sea planning. Some studies use dynamic hydrological and biogeochemical models to capture processes such as variation in sediment run-off across different land uses and storage of sediments behind dams (e.g. Álvarez-Romero *et al.* 2015), whereas other studies have used simpler empirical models that are static representations of run-off (e.g. Brown *et al.* 2017a).

Empirical-based models have been popular in existing land-sea plans, because they can easily be developed using GIS software and applied to regions with little local data. Empirical-based models are a simplified representation of natural processes based on field observations of catchment processes. They are frequently used in modeling complex processes and are particularly useful for identifying the sources of sediments within a catchment. For example, the INVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) tool, a package of tools for ecosystem service evaluation uses sediment delivery ratio, which is an empirical model that relates sediment yield to flow discharge (Hamel *et al.* 2015).

A major challenge for empirical-based models is meaningful parameterization across regions with limited data. Model parameters that are borrowed from other regions may misestimate actual pollutant loads by orders of magnitude (Hamel *et al.* 2017). A recent analysis found that global-scale models of sediment delivery were highly inaccurate when compared against local data, however predictions could be drastically improved by accounting for local spatial variability in erosion and sediment delivery, even with limited local scale data (Hamel *et al.* 2017). The approach offers a way forward for developing relatively accurate empirical methods that can estimate sediment delivery in catchments that lack gauge data.

Some recent land-sea plans have used dynamic hydrological and biogeochemical models and found that the additional investment in model complexity can vastly change recommendations for planning priorities. For instance, a comparison of two physical models found that sediment loadings may be over-predicted by models that do not account for storage of sediment in reservoirs (Álvarez-Romero *et al.* 2014). Further, models based on physical processes may often capture processes not accounted for in empirical models. For instance, two primary sources of sediment erosion are from hill-slopes and gullies, but most empirical models account only for hill-slope erosion. In some catchments, erosion from gullies can be the primary source of sediment, so models that ignore gully erosion may miss the importance of stream-side vegetation for averting sediment loss (Olley *et al.* 2015).

Dispersion and transformation of pollutants in the ocean

A major challenge with modelling of water quality is ensuring the parameters being quantified are relevant to marine ecosystems and fisheries. For instance, numerous planning studies have modelled pollution using indices of cumulative threat. However, the ecological response can depend strongly on the units of the water quality variable. For instance, various components of water quality are dispersed and processed in coastal waters in very different ways. Fine silts and clays will stay suspended for longer periods of time and thus may disperse further than coarse sediment, which will tend to settle out sooner (Bainbridge *et al.* 2014). Most geographies that face water quality issues will be affected by multiple pollutants, for instance, sedimentation and nutrient loads are typically correlated in an agricultural setting, yet may act independently in urban point source environments. Multiple models of dispersal may need to be employed to accurately predict the dispersal of multiple components.

Land-sea plans have primarily used three types of approaches to model the dispersion of pollutants in the ocean. The first is to model gradients in water quality on a nominal 'threat-scale' that putatively represents a gradient of impacts to marine ecosystems (Halpern *et al.* 2009). Threat is often dispersed from rivers to ocean ecosystems using a simple distance decay function (e.g. Giakoumi *et al.* 2015). The second approach is to explicitly model one aspect of water quality, like sedimentation, using a combination of geographic information systems and some simple process models (e.g. Rude *et al.* 2015). The third approach is hydrodynamic modelling to link river outflow to ocean water quality (e.g. Paris & Chérubin 2008). However, none of the planning studies (Table S1) or modelling studies (Table 1) we reviewed have used hydrodynamic models to explicitly model ocean processes that disperse and transform pollutants. An exception is the recently developed, high-resolution model suite eReefs, which with further development will link water-quality change in receiving waters to land-use change (https://ewater.org.au/products/ewater-source/).

Modelling dispersal of pollutants in a broader range of regions requires simple models that can operate effectively in data limited regions. Models that combine GIS tools for calculating source to reef distances with simple current models may be more useful than complex hydrodynamic models in data poor regions. For instance, sediment exposure of reefs in Indonesia was modelled using this simple approach (Rude *et al.* 2015) and sediment and nutrient exposure on the Great Barrier Reef were modelled using GIS tools (Maughan and Brodie 2009). Further, work is needed to empirically validate these simple models, at least in some regions, against in situ water quality measurements and, ideally, time integrated measurements like coral cores (Maina *et al.* 2012). For instance, Bayesian models have been used to verify GIS based approaches to modelling sediment dispersion against satellite and in-situ data of reefs (Brown *et al.* 2017a).

Models of the dispersal of pollutants in the ocean are typically complex and are built for specific case studies where detailed bathymetric, tidal, hydrodynamic data are available and, as such, their application in data poor regions is usually not feasible. For instance, three dimensional ocean circulation models, previously used to model dispersal of fish larvae, have been adapted to modelling sediment dispersion in the Caribbean (Paris & Chérubin 2008). However, this model was based on a regional ocean model with a horizontal resolution of 2 km and 25 depth layers. Global ocean current model have low resolution (Chassignet *et al.* 2007) that does not adequately resolve coastal currents. Small-scale hydrological drivers, including tides and winds, can also be important determinants of coastal water quality (e.g. Golbuu *et al.* 2011). The re-suspension of sediments by currents, wind and waves can see the impacts of run-off occurring for many decades past the time of input (Fabricius *et al.* 2014).

Response of ecosystems and fish populations to pollution

Modelling the effects of pollutants on fisheries is challenging due to numerous potential mechanisms for pollutants to affect fish populations. Pollution can affect individual fish directly, for instance, sewage pollution can increase the rate of pathologies in fish gills and livers (Schlacher, Mondon & Connolly 2007). Pollution can also affect fish populations indirectly by altering their interactions with other organisms and their habitats (Brown *et al.* 2017b). Indirect effects of pollution on fish can be broadly classed into mediation and moderation effects. Mediation of pollution occurs when pollution affects a component of the ecosystem on which the fish is directly dependent. For instance, turbidity can cause declines in coral on which butterflyfish feed, thus potentially causing declines in these species (Brown *et al.* 2017b). Moderation effects occur when pollution affects a population's interactions with other populations. For instance, turbid waters may impede a predatory fish's ability to find prey items (Wenger *et al.* 2013).

The challenge for modelling the effects of pollution on fish populations is choosing a model for the types of mechanisms that most likely operate in the planning context. For instance, direct physiological effects may be the dominant driver of fish population declines if the primary pollutant is sewage, whereas indirect moderation effects may be important if sediment pollution affects water clarity and foraging ability. In general, modelling of ecosystems responses to pollution follow one of three approaches: (1) simple habitat area relationships; (2) empirical (often statistical models); or (3) simulation modelling.

Habitat area models typically work on the assumption of a mediating effect of habitats on fish populations. It is assumed that fish populations will change in proportion to the area of habitat that is lost or gained. An issue with assuming that habitat area relates to fishery production is that such relationships have often failed to hold up empirically (Sheaves *et al.* 2014). Habitat area relationships may fail, because they do not account for ecological interactions among multiple components of water quality. For instance, increasing nutrient loads may improve food for fish and mask declines in vegetative habitats (Sheaves *et al.* 2014).

Empirical models have been broadly used to try to quantify the impact of pollution on fish populations and learn about the mechanisms through which pollution affects fish populations. Statistical models can leverage either spatial or temporal variability in water quality to both contrast gradients in fish abundance (or biomass) across gradients in water quality. For instance, abundances of juvenile flatfish in their nursery habitats around Elkhorn Slough (California) are lower in years when dissolved oxygen is lower in their nursery grounds (Hughes *et al.* 2015). Hypoxia may reduce the available area suitable for recruitment of juvenile sole and sanddab, thus ultimately reducing adult abundance and fishery catch in the years following hypoxic conditions (Hughes *et al.* 2015).

A challenge for many fisheries is obtaining adequate time-series data to provide the sample size across years with both poor and high water quality. Further, many of the responses of fish populations to changing water quality may lag years or occur as cumulative exposure. One solution to identifying water quality impacts where there are poor temporal contrasts is to undertake field studies that seek to identify the processes of connections. For instance, in the Nile River Delta, stable isotopes were used to detect high quantities of sewage-derived nutrients in a fishery (Oczkowski *et al.* 2009). This additional process detail provided context to observed changes in catch and revealed that sewage had enhanced production in this system.

Spatial comparisons may be more feasible in many regions, although they require concerted field efforts to obtain adequate data and also suitable control sites. In Chile, a large number of watersheds with varying land-uses enabled comparison of shellfish health under different pressures levels and show that shellfish health was poorer nearer degraded watersheds (Van Holt *et al.* 2012). A challenge with spatial comparisons is identifying appropriate control sites. For instance, turbidity is often higher nearer to shore, creating a gradient that can be used to correlate turbidity with fish abundance or community structure. However, the inshoreoffshore gradient may also be correlated with other drivers of fish communities, such as exposure to waves (Delevaux *et al.* 2018).

Simulations of fish responses to water quality have commonly been used to evaluate the outcomes of different management strategies (Weijerman, Fulton & Brainard 2016). Simulation models are typically parameterised using a priori evidence for the processes linking water quality and fish populations and limited local time-series data. For instance, an ecosystem model that was developed for coral reefs of Guam and was used to assess the interactive effects of management of fisheries and land-based sources of pollution on indicators of fishery health (Weijerman, Fulton & Brainard 2016). The model included mechanistic descriptions for the response of benthic ecosystems to water pollution (nutrients and sedimentation). Scenario modelling found that removal of land-based sources of pollution had considerable benefits for fisheries landings. However, landings were compromised with improved water quality and strict fisheries regulations. The model was informative because it demonstrated how a compromise among different objectives could be achieved through simultaneous management of land-based sources of pollution and fisheries.

Economic, social and human responses to ecological change

Economic and social context play an important role in determining the impact of land-uses to fisheries, though we found only three examples of quantitative land-sea models that considered socio-economic responses (Table 1). Coastal fisheries may depend heavily on particular species, and the response of those species to run-off will be most important for the economy, but not necessarily the ecosystem. For instance, loco (*Concholepas concholepas*) are a highly valued mollusc from Chile's coast, but loco in coastal areas that are affected by run-off from tree plantations are of lower quality and gain lower prices at market (Van Holt *et al.* 2012). Thus, tree plantations have had a significant effect on people who derive their livelihood from loco fishing (Van Holt 2012).

There are few examples of socio-economic models for the impacts of water quality; however, insights can be gained from the marine reserve literature. Early work on marine reserves had a biological focus and was criticised for unrealistic socio-economic dynamics, in particular, they ignored the displacement of fishing pressure and that can potentially increase fishing pressure in non-reserved areas (Mascia & Claus 2009). Behavioural responses to changes in water quality should also be considered when developing land-sea plans.

For instance, if a fishery declines because of poor water quality, fishers switch to other fisheries (Van Holt 2012). Adaptive responses may mitigate the impact of land-uses on the affected livelihoods, but also increase pressure on the alternative fisheries (Van Holt 2012). The adaptive capacity of fishers needs to consider social factors, like fishing experience, income and age (Van Holt 2012). In Chile, fishers who left the loco fishery due to poor water quality moved to a different fishery if they were experienced in fishing, but sought alternative employment if there were an inexperienced fisher (Van Holt 2012). Poverty traps, where fishers that are invested in exploiting an degraded resource cannot afford to switch careers, are a risk for fishers with low adaptive capacity (Cinner, Daw & McClanahan 2009), and will magnify the impacts of land-uses.

Socio-economic models of land-use impacts to fisheries have tended to focus on the economic aspects. For instance, time-series of salmon production across different watersheds with different levels of land-use change allowed estimation of the per hectare cost of land-use change on salmon fisheries (Knowler *et al.* 2003). Such estimates provide economic justification for spending on restoration of habitats. Similarly, the effects of agricultural run-off causing hypoxia suggest economically optimal fishery management for brown shrimp needs to account for hypoxia (Huang & Smith 2011).

A further barrier to land-sea planning is coordinating management actions across different jurisdictions. In many countries different governmental institutions manage land-use and ocean management and they make lack incentives for coordinating their actions. Land-sea management may often require the coordination of different land-holders. For instance, in West Maui, Hawaii, degraded roads on the land of just a few landholders contribute the majority of sediment run-off to coastal marine ecosystems (Oleson *et al.* 2017). Much greater reductions in sediment may be achieved if road repair is prioritised and focussed on just the few most cost-effective roads for repair, rather than sharing resources for road repair across all land-holders (Oleson *et al.* 2017).

Conclusion and future directions

The strong linkages between coastal fisheries and terrestrial run-off demand that marine resource management evolve to consider human activities on land. We have reviewed the processes that link the drivers of land-use change to management responses required to sustain coastal ecosystem services. The complexity of processes linking land-use change to change in coastal hinders effective integrated land-sea planning. Overcoming this complexity can be facilitated through efforts to integrate models from the drivers of land-use change to management responses for marine ecosystems. Based on our review, we suggest several future research directions for connecting land and sea models that will assist integrated land-sea planning:

1. Where possible, researchers should attempt to integrate simple static tools for predicting impacts of land-use change on coastal fisheries (e.g. Brown *et al.* 2017a;Brown *et al.* 2017b), instead of using threat indices. The advantages of static models are that they are easily deployed and parameterised by small teams, but can be used to evaluate objectives in terms of outcomes (like fishery yield), rather than threat. Toolboxes like INVEST (Hamel *et al.* 2017) promise to aid in this challenge, but more work is needed to evaluate their accuracy across a range of linked fishery-catchment types.

2. A gap in existing models of land-sea processes is the consideration of how large-scale drivers, like climate change and globalisation of economies, impact on the effectiveness of land-sea plans (Table 1).
3. A challenge for planning with static or dynamic models is the consideration of uncertainty at any stage of the linked land-sea process. Methods are needed that can propagate uncertainty, so that the key uncertainties can be quantified, such as the use of Bayesian modelling techniques to model uncertainty in the extent of sediment run-off impacts to marine ecosystems (Brown *et al.* 2017a;Brown & Hamilton 2018).
4. The effects of extreme weather events on run-off and coastal fisheries are poorly understood. More work is needed to understand how extreme events affect coastal fisheries indirectly by temporary changes in water quality, and how the timing and severity of events may change under different climate change scenarios.
5. Models that can consider dynamic feedbacks in socio-economic systems like fisher behavioural responses to changes in water quality are needed. Dynamic feedbacks may render plans ineffective or may be supported by planners where they enhance the capability of people to adapt to changes in fisheries (e.g. Van Holt 2012).
6. Management plans are often developed on relatively short-time scales with limited funds for future research. To this ends, robust guidelines for planning, such as rules of thumb that can be used to inform planning in data-poor situations are needed. For instance, geographic context can be used to decide whether actions on the land or in the sea are more cost effective for achieving conservation of marine habitats (Saunders et al. 2017a). Similar rules of thumb are needed for the socio-economic impacts of run-off.
7. Actively involving a wide range of stakeholders (such as industry, local fishers, NGOs, government departments in planning processes is a fundamental step in integrated land -sea management. This is particularly important in data poor regions, where

participatory mapping exercises have proven to be highly effective at extracting fine scale spatial information on current and future land based threats to marine systems (e.g. Game *et al.* 2011).

8. While environmental NGOs may have the skills and relationships needed to facilitate participatory planning processes in data poor countries, they rarely have the expertise needed to develop dynamical models that link land to sea processes. Our experiences demonstrate that one way to address this gap is to have university based ecological modellers become stakeholders in planning processes.

Finally, the complexity of comprehensive modelling of linked land-sea processes should not hold back the development of management plans. A pragmatic way to proceed in the absence of planning tools that account for land-sea impacts is to devise plans using expert input and then evaluate ecological and socio-economic outcomes post-hoc using existing modelling tools.

Quantitative planning for the impacts of land-use change on coastal fisheries requires linking models across a multitude of disciplines. Doing so can be a challenge for the small teams often tasked with developing land-sea plans. Addressing the research challenges outlined above should help those teams develop plans that focus on outcomes, like fish yield, rather than more abstract objectives of reducing threat. Outcome-driven planning is likely to be more effective for driving land-sea plans and evaluating competing trade-offs.

Author contributions

CB and SJ wrote the first draft. All authors contributed to subsequent drafts.

Acknowledgements

CJB was supported by a Discovery Early Career Researcher Award (DE160101207) from the Australian Research Council. Funding for workshops to develop this paper was provided by the Science for Nature and People Partnership (SNAPP) to the Ridges to Reef Fisheries Working Group. SNAPP is a collaboration of The Nature Conservancy, the Wildlife Conservation Society and the National Center for Ecological Analysis and Synthesis (NCEAS).

References

Álvarez-Romero, J.G., Pressey, R.L., Ban, N.C. & Brodie, J. (2015) Advancing Land-Sea Conservation Planning: Integrating Modelling of Catchments, Land-Use Change, and River Plumes to Prioritise Catchment Management and Protection. *Plos One,* **10**

Alvarez-Romero, J.G., Pressey, R.L., Ban, N.C., Vance-Borland, K., Willer, C., Klein, C.J. & Gaines, S.D. (2011) Integrated land-sea conservation planning: the missing links. *Annual Review of Ecology, Evolution, and Systematics,* **42,** 381-409.

Álvarez-Romero, J.G., Wilkinson, S.N., Pressey, R.L., Ban, N.C., Kool, J. & Brodie, J. (2014) Modeling catchment nutrients and sediment loads to inform regional management of water quality in coastal-marine ecosystems: A comparison of two approaches. *Journal of Environmental Management,* **146,** 164-178.

Arkema, K.K., Verutes, G.M., Wood, S.A., Clarke-Samuels, C., Rosado, S., Canto, M., Rosenthal, A., Ruckelshaus, M., Guannel, G. & Toft, J. (2015) Embedding ecosystem services in coastal planning leads to better outcomes for people and nature. *Proceedings of the National Academy of Sciences,* **112,** 7390-7395.

Bainbridge, Z.T., Lewis, S.E., Smithers, S.G., Kuhnert, P.M., Henderson, B.L. & Brodie, J.E. (2014) Fine - suspended sediment and water budgets for a large, seasonally dry tropical catchment: Burdekin River catchment, Queensland, Australia. *Water Resources Research,* **50,** 9067-9087.

Bohensky, E., Butler, J.R., Costanza, R., Bohnet, I., Delisle, A., Fabricius, K., Gooch, M., Kubiszewski, I., Lukacs, G. & Pert, P. (2011) Future makers or future takers? A scenario analysis of climate change and the Great Barrier Reef. *Global Environmental Change,* **21,** 876-893.

Brown, C.J. & Hamilton, R. (2018) Estimating the footprint of pollution on coral reefs with models of species turnover. *Conservation Biology,*

Brown, C.J., Jupiter, S.D., Albert, S., Klein, C.J., Mangubhai, S., Maina, J.M., Mumby, P.J., Olley, J., Stewart-Koster, B., Tulloch, V.J. & Wenger, A.S. (2017a) Tracing the influence of land-use change on coral reefs using a Bayesian model. *Scientific Reports,* **7,** 4740.

Brown, C.J., Jupiter, S.D., Lin, H.-Y., Albert, S., Klein, C., Maina, J.M., Tulloch, V.J., Wenger, A.S. & Mumby, P.J. (2017b) Habitat change mediates the response of coral reef fish populations to terrestrial run-off. *Marine Ecology Progress Series,* **576,** 55-68.

Carroll, C., Waters, D., Vardy, S., Silburn, D.M., Attard, S., Thorburn, P.J., Davis, A.M., Halpin, N., Schmidt, M. & Wilson, B. (2012) A paddock to reef monitoring and modelling framework for the Great Barrier Reef: paddock and catchment component. *Marine Pollution Bulletin,* **65,** 136-149.

Castiblanco, C., Etter, A. & Aide, T.M. (2013) Oil palm plantations in Colombia: a model of future expansion. *Environmental Science & Policy,* **27,** 172-183.

Chassignet, E.P., Hurlburt, H.E., Smedstad, O.M., Halliwell, G.R., Hogan, P.J., Wallcraft, A.J., Baraille, R. & Bleck, R. (2007) The HYCOM (hybrid coordinate ocean model) data assimilative system. *Journal of Marine Systems,* **65,** 60-83.

Christie, P., Lowry, K., White, A.T., Oracion, E.G., Sievanen, L., Pomeroy, R.S., Pollnac, R.B., Patlis, J.M. & Eisma, R.-L.V. (2005) Key findings from a multidisciplinary examination of integrated coastal management process sustainability. *Ocean & Coastal Management,* **48,** 468-483. Available from http://www.sciencedirect.com/science/article/pii/S0964569105000608

Cinner, J., Daw, T. & McClanahan, T. (2009) Socioeconomic factors that affect artisanal fishers' readiness to exit a declining fishery. *Conservation Biology,* **23,** 124-130.

Comte, I., Colin, F., Whalen, J.K., Grünberger, O. & Caliman, J.-P. (2012) Agricultural practices in oil palm plantations and their impact on hydrological changes, nutrient fluxes and water quality in Indonesia: a review. *Advances in Agronomy*, pp. 71-124. Elsevier,

Delevaux, J.M., Whittier, R., Stamoulis, K.A., Bremer, L.L., Jupiter, S., Friedlander, A.M., Poti, M., Guannel, G., Kurashima, N. & Winter, K.B. (2018) A linked land-sea modeling framework to inform ridge-to-reef management in high oceanic islands. *Plos One,* **13,** e0193230.

Fabricius, K., Logan, M., Weeks, S. & Brodie, J. (2014) The effects of river run-off on water clarity across the central Great Barrier Reef. *Marine Pollution Bulletin,* **84,** 191-200.

Fitzherbert, E.B., Struebig, M.J., Morel, A., Danielsen, F., Brühl, C.A., Donald, P.F. & Phalan, B. (2008) How will oil palm expansion affect biodiversity? *Trends in ecology & evolution,* **23,** 538-545.

Game, E.T., Kareiva, P. & Possingham, H.P. (2013) Six common mistakes in conservation priority setting. *Conservation Biology,* **27,** 480-485.

Game, E.T., Lipsett - Moore, G., Hamilton, R., Peterson, N., Kereseka, J., Atu, W., Watts, M. & Possingham, H. (2011) Informed opportunism for conservation planning in the Solomon Islands. *Conservation Letters,* **4,** 38-46.

Giakoumi, S., Brown, C.J., Katsanevakis, S., Saunders, M.I. & Possingham, H.P. (2015) Using threat maps for cost-effective prioritization of actions to conserve coastal habitats. *Marine Policy,* **61,** 95-102.

Golbuu, Y., Wolanski, E., Harrison, P., Richmond, R.H., Victor, S. & Fabricius, K.E. (2011) Effects of land-use change on characteristics and dynamics of watershed discharges in Babeldaob, Palau, Micronesia. *Journal of Marine Biology,* **2011**

Halpern, B.S., Ebert, C.M., Kappel, C.V., Madin, E.M., Micheli, F., Perry, M., Selkoe, K.A. & Walbridge, S. (2009) Global priority areas for incorporating land–sea connections in marine conservation. *Conservation Letters,* **2,** 189-196.

Hamel, P., Chaplin-Kramer, R., Sim, S. & Mueller, C. (2015) A new approach to modeling the sediment retention service (InVEST 3.0): Case study of the Cape Fear catchment, North Carolina, USA. *Science of The Total Environment,* **524,** 166-177.

Hamel, P., Falinski, K., Sharp, R., Auerbach, D.A., Sánchez-Canales, M. & Dennedy-Frank, P.J. (2017) Sediment delivery modeling in practice: Comparing the effects of watershed characteristics and data resolution across hydroclimatic regions. *Science of The Total Environment,* **580,** 1381-1388.

Hilborn, R. & Walters, C.J. (2013) *Quantitative fisheries stock assessment: choice, dynamics and uncertainty*. Springer Science & Business Media,

Huang, L. & Smith, M.D. (2011) Management of an annual fishery in the presence of ecological stress: the case of shrimp and hypoxia. *Ecological Economics,* **70,** 688-697.

Hughes, B.B., Levey, M.D., Fountain, M.C., Carlisle, A.B., Chavez, F.P. & Gleason, M.G. (2015) Climate mediates hypoxic stress on fish diversity and nursery function at the land–sea interface. *Proceedings of the National Academy of Sciences,* **112,** 8025-8030.

Klein, C.J., Ban, N.C., Halpern, B.S., Beger, M., Game, E.T., Grantham, H.S., Green, A., Klein, T.J., Kininmonth, S. & Treml, E. (2010) Prioritizing land and sea conservation investments to protect coral reefs. *Plos One,* **5,** e12431.

Klein, C.J., Jupiter, S.D., Selig, E.R., Watts, M.E., Halpern, B.S., Kamal, M., Roelfsema, C. & Possingham, H.P. (2012) Forest conservation delivers highly variable coral reef conservation outcomes. *Ecological Applications,* **22,** 1246-1256.

Knowler, D.J., MacGregor, B.W., Bradford, M.J. & Peterman, R.M. (2003) Valuing freshwater salmon habitat on the west coast of Canada. *Journal of Environmental Management,* **69,** 261-273.

Lambin, E.F. & Meyfroidt, P. (2011) Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences,* **108,** 3465-3472.
Maina, J., de Moel, H., Vermaat, J.E., Bruggemann, J.H., Guillaume, M.M., Grove, C.A., Madin, J.S., Mertz-Kraus, R. & Zinke, J. (2012) Linking coral river runoff proxies with climate variability, hydrology and land-use in Madagascar catchments. *Marine Pollution Bulletin,* **64,** 2047-2059.
Mascia, M.B. & Claus, C. (2009) A property rights approach to understanding human displacement from protected areas: the case of marine protected areas. *Conservation Biology,* **23,** 16-23.
Melbourne-Thomas, J., Johnson, C., Alino, P., Geronimo, R.C., Villanoy, C. & Gurney, G. (2011) A multi-scale biophysical model to inform regional management of coral reefs in the western Philippines and South China Sea. *Environmental Modelling & Software,* **26,** 66-82.
Oczkowski, A.J., Nixon, S.W., Granger, S.L., El-Sayed, A.-F.M. & McKinney, R.A. (2009) Anthropogenic enhancement of Egypt's Mediterranean fishery. *Proceedings of the National Academy of Sciences,* **106,** 1364-1367.
Oleson, K.L., Falinski, K.A., Lecky, J., Rowe, C., Kappel, C.V., Selkoe, K.A. & White, C. (2017) Upstream solutions to coral reef conservation: The payoffs of smart and cooperative decision-making. *Journal of Environmental Management,* **191,** 8-18.
Olley, J., Burton, J., Hermoso, V., Smolders, K., McMahon, J., Thomson, B. & Watkinson, A. (2015) Remnant riparian vegetation, sediment and nutrient loads, and river rehabilitation in subtropical Australia. *Hydrological Processes,* **29,** 2290-2300.
Paris, C. & Chérubin, L. (2008) River-reef connectivity in the Meso-American Region. *Coral Reefs,* **27,** 773-781.
Rude, J., Minks, A., Doheny, B., Tyner, M., Maher, K., Huffard, C., Hidayat, N.I. & Grantham, H. (2015) Ridge to reef modelling for use within land–sea planning under data - limited conditions. *Aquatic Conservation: Marine and Freshwater Ecosystems,*
Saunders, M.I., Atkinson, S., Klein, C.J., Weber, T. & Possingham, H.P. (2017a) Increased sediment loads cause non-linear decreases in seagrass suitable habitat extent. *Plos One,* **12,** e0187284.
Saunders, M.I., Bode, M., Atkinson, S., Klein, C.J., Metaxas, A., Beher, J., Beger, M., Mills, M., Giakoumi, S. & Tulloch, V. (2017b) Simple rules can guide whether land-or ocean-based conservation will best benefit marine ecosystems. *PLoS Biology,* **15,** e2001886.
Schlacher, T.A., Mondon, J.A. & Connolly, R.M. (2007) Estuarine fish health assessment: evidence of wastewater impacts based on nitrogen isotopes and histopathology. *Marine Pollution Bulletin,* **54,** 1762-1776.
Selkoe, K.A., Blenckner, T., Caldwell, M.R., Crowder, L.B., Erickson, A.L., Essington, T.E., Estes, J.A., Fujita, R.M., Halpern, B.S. & Hunsicker, M.E. (2015) Principles for managing marine ecosystems prone to tipping points. *Ecosystem Health and Sustainability,* **1,** 1-18.
Sheaves, M., Brookes, J., Coles, R., Freckelton, M., Groves, P., Johnston, R. & Winberg, P. (2014) Repair and revitalisation of Australia′ s tropical estuaries and coastal wetlands: Opportunities and constraints for the reinstatement of lost function and productivity. *Marine Policy,* **47,** 23-38.
Stoms, D.M., Davis, F.W., Andelman, S.J., Carr, M.H., Gaines, S.D., Halpern, B.S., Hoenicke, R., Leibowitz, S.G., Leydecker, A. & Madin, E.M. (2005) Integrated coastal reserve planning: making the land–sea connection. *Frontiers in Ecology and the Environment,* **3,** 429-436.
Tulloch, V.J., Brown, C.J., Possingham, H.P., Jupiter, S.D., Maina, J.M. & Klein, C. (2016) Improving conservation outcomes for coral reefs affected by future oil palm development in Papua New Guinea. *Biological Conservation,* **203,** 43-54.

Van Holt, T. (2012) Landscape influences on fisher success: adaptation strategies in closed and open access fisheries in southern chile. *Ecology and Society,* **17,** 28.

Van Holt, T., Moreno, C.A., Binford, M.W., Portier, K.M., Mulsow, S. & Frazer, T.K. (2012) Influence of landscape change on nearshore fisheries in southern Chile. *Global Change Biology,* **18,** 2147-2160.

Weijerman, M., Fulton, E.A. & Brainard, R.E. (2016) Management strategy evaluation applied to coral reef ecosystems in support of ecosystem-based management. *Plos One,* **11,** e0152577.

Wenger, A., McCormick, M., McLeod, I. & Jones, G. (2013) Suspended sediment alters predator–prey interactions between two coral reef fishes. *Coral Reefs,* **32,** 369-374.