RESTORATION ECOLOGY
The Journal of the Society for Ecological Restoration

RESEARCH ARTICLE

Applying systematic conservation planning to improve the allocation of restoration actions at multiple spatial scales

Ben L. Gilby[1,2], Andrew D. Olds[1], Christopher J. Brown[3], Rod M. Connolly[4], Christopher J. Henderson[1], Paul S. Maxwell[5,6], Thomas A. Schlacher[1]

Ecological restoration is increasingly being upscaled to larger spatial scales of tens to hundreds of kilometers. Yet the complex logistics and high costs of ecological restoration mean that actions must be placed strategically at local scales of tens of meters to maximize ecological benefits and reduce socioeconomic costs. Despite the purported use of systematic planning tools for allocating restoration effort, the uptake and implementation of data-driven restoration planning and ecological goal setting remains poor in many restoration programs. Here we demonstrate how the sequential workflows of systematic conservation planning can be translated to restoration at two spatial scales to enhance estuarine fisheries in eastern Australia. We select estuaries where restoration is feasible and recommended based on quantitative regional ecological goals (i.e. regional-scale prioritization), and then identify potential restoration sites at smaller spatial scales within estuaries based on the principles of spatial ecology to ensure that the success and benefits of restoration are maximized (i.e. local-scale prioritization). At the regional scale, we identified four levels of restoration priorities (very high, high, intermediate, and low) using quantitative ecological goals and the current ecological understanding of each system. At the local scale, we used spatially explicit Bayesian belief networks to identify sites that maximize restoration outcomes based on the environmental niche of habitat-forming species and the spatial configuration of habitats that maximizes their use by fish. We show that using systematic frameworks can become an essential tool to optimize restoration investments at multiple scales as efforts upscale globally.

Key words: coastal ecosystems, feasibility, mangroves, oyster reefs, seagrass, spatial planning

Implications for Practice

- As restoration actions expand in spatial extent and scope, we need better strategies to prioritize actions.
- Systematically prioritizing effort at broad scales, and then selecting suitable restoration sites at narrower will maximize the effectiveness and efficiency of broad restoration programs.
- Using data to identify these priorities must become a hallmark of landscape-scale restoration.

Introduction

Several international agreements mandate that ecosystems be restored at regional scales of tens to hundreds of kilometers (e.g. Tobon et al. 2017). Restoration is usually applied for reestablishing or rehabilitating habitats at small spatial scales (10s to 100s m) but planning for regional-scale restoration necessitates broader spatial perspective (10s to 100s km) (Neeson et al. 2015; Roy et al. 2018). As the spatial extent of restoration increases, the challenge is to employ and prioritize restoration actions to achieve regional-scale restoration goals but implement these actions strategically at smaller spatial scales to ensure that sites chosen for restoration can support habitat-forming species and maximize ecological benefits (Wilson et al. 2011; Cattarino et al. 2018). Consequently, there is a need to apply more rigorous decision-making processes for restoration actions at both regional and local scales to maximize project feasibility and ecological benefits (Adame et al. 2015; Cattarino et al. 2016; Shoo et al. 2017).

Complex restoration decisions could become more structured and transparent by using the stepwise workflow of systematic conservation planning (Possingham et al. 2000). Restoration plans increasingly use components of systematic conservation planning for allocating restoration effort (Langhans

Author contributions: BLG, ADO, RMC, PSM, TAS conceived and designed the research; BLG, CJH undertook the field work and analysis; all authors contributed to interpreting patterns and writing and editing the manuscript.

[1]School of Science and Engineering, University of the Sunshine Coast, Maroochydore DC, Queensland 4558, Australia
[2]Address correspondence to B. L. Gilby, email bgilby@usc.edu.au
[3]Australian Rivers Institute, Griffith University, Nathan, Queensland 4111, Australia
[4]Australian Rivers Institute—Coasts and Estuaries, School of Environment and Science, Griffith University, Gold Coast, Queensland 4222, Australia
[5]Healthy Land and Water, Level 4, 200 Creek Street, Spring Hill, Queensland 4004, Australia
[6]Alluvium Consulting, Suite 14, 36 Agnes St, Fortitude Valley, Queensland 4006, Australia

doi: 10.1111/rec.13403
Supporting information at:
http://onlinelibrary.wiley.com/doi/10.1111/rec.13403/suppinfo

et al. 2016), and studies presenting these frameworks support their use for restoration (Margules & Pressey 2000). The framework first proposed by Margules and Pressey (2000) (and then expanded on elsewhere; e.g. Pressey & Bottrill 2009) presents sequential stages for the selection principally of reserves (Fig. 1). While each of these stages can be used to help prioritize restoration effort and might be helpful as the spatial scale and extent of restoration increases, it appears that uptake of systematic conservation planning methods for restoration remains rare. For example, reviews of restoration planning indicate ongoing issues with quantitative goal setting (Thorpe & Stanley 2011; Hallett et al. 2013) and a lack of strategic placement of restoration sites (Gilby et al. 2018a). Quantitative objectives are important because they show how existing habitat values and actions contribute toward current-day values within planning units, and provide a level to which restoration needs to be completed to reach objectives (Margules & Pressey 2000). Creating thorough, quantitatively optimized and defensible restoration plans and goals (i.e. the condition or state of that ecosystem and attributes that the project is aiming to achieve; Gann et al. 2019) via processes such as systematic conservation planning should be an important focus for practitioners as the desire to restore whole landscapes increases (Thompson 2011).

Ecological restoration actions are prioritized across landscapes at multiple spatial scales. At the regional scale, decisions are first made as to whether restoration will be more effective than protecting existing habitats (i.e. in reserves) or applying other intervention strategies (like heightened legislation) (Saunders et al. 2017; Cattarino et al. 2018). Required extents of restoration can then be applied regionally based on a quantitative understanding of historical habitat extent (Higgs et al. 2014) or key landscape processes, ecological functions, or ecosystem service recovery (Simenstad et al. 2006; Hermoso et al. 2021). Re-creating past attributes is, however, problematic due to sliding baselines, lost or inaccurate historical information, and long-acting human stressors (e.g. urbanization, climate change, invasive species) that have created landscapes and biotic assemblages that are fundamentally different from their historical predecessors (Thorpe & Stanley 2011; Balaguer et al. 2014). The primary goal of ecological restoration is to restore self-sustaining and self-organizing ecosystems that are integrated within their broader landscape (Gann et al. 2019). Therefore, regional restoration extent goals should usually be set at data-informed levels of (1) key landscape processes (i.e. hydrological, geomorphological, or other physical processes that influence ecosystem structure at landscape scales); (2) ecological functions (i.e. of or related to the movement or storage of energy or nutrients); or (3) the rebound in ecosystem services produced by the restoration investment (Simenstad et al. 2006). At the local scale, restoration actions must be placed strategically according to small-scale environmental factors (<10 m) to maximize the growth and persistence of the restored ecosystem (especially of habitat-forming species) and potential ecological benefits (Shoo et al. 2017; Gilby et al. 2019*a*). This second, narrower scale of identifying suitable sites for

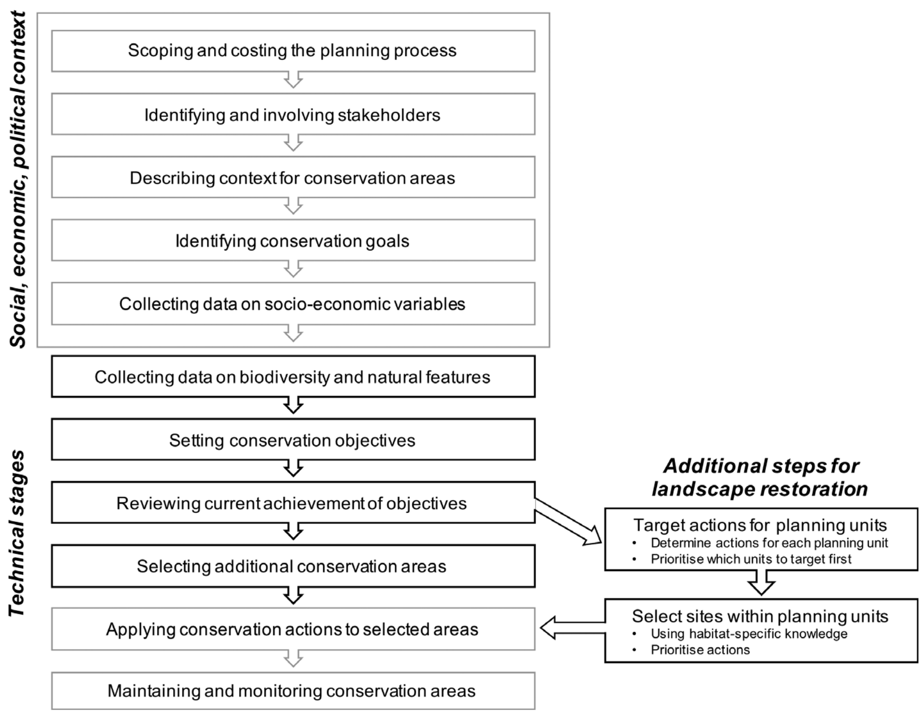

Figure 1. Flowchart of stages for systematic conservation planning and suggested additions to this framework to assist in the prioritization of restoration efforts. Adapted from Pressey and Bottrill (2009).

restoration is arguably a more important attribute of restoration planning and prioritization than for reserves because the implementation of restoration is more expensive per unit area (Bayraktarov et al. 2016), and because misplacing a restoration site by even meters might result in restoration failure (Gilby et al. 2019*a*).

As the spatial extent of restoration expands and the need to place restoration efforts strategically across landscapes increases, practitioners require more thorough, quantitative, and defensible frameworks to allocate limited resources. Restoration projects also need to more thoroughly integrate the principles of quantitative ecological goal setting (Thorpe & Stanley 2011; Hallett et al. 2013; Gann et al. 2019) and spatial prioritization (Gilby et al. 2018a; Hermoso et al. 2021) into their planning. Despite consistent references to the application of systematic conservation planning to restoration, perhaps due to the incorrect assumption that these sorts of frameworks are regularly used for restoration on the ground, there are few worked examples in the literature that can be used as models by practitioners to assist in decision-making. The objective of this study is to address the challenges of planning for ecological restoration at multiple spatial scales, and to thoroughly incorporate quantitative and spatially explicit goal setting by stepping through the process of systematic conservation planning to prioritize and place restoration efforts at multiple scales across landscapes (Fig. 1). We show that using systematic frameworks can become an essential tool to optimize restoration investments at multiple scales as efforts upscale globally. These frameworks provide multiple opportunities to incorporate key data and ensure that quantitative objectives are set for the restoration actions.

Methods

Conceptual and Analytical Approach

We demonstrate the utility of a systematic conservation planning framework for restoration by presenting a case study to prioritize seagrass, oyster reef, and mangrove restoration undertaken to improve habitat values for fish and fisheries in 13 estuaries in southeast Queensland (SEQ), eastern Australia. Here, we consider a regional manager who must prioritize the restoration of estuarine habitats at a regional scale across estuaries (10s to 100s km) to achieve broad goals for habitat recovery, but also identify restoration sites at local scales within estuaries (<100 s m) to maximize the potential benefits of restoration for both habitat recovery and key ecosystem services.

We collect information on the current-day distribution of fish in estuaries using underwater videography approaches and gather information on the current-day distribution of marine habitats from local governments. We set quantitative regional restoration goals at the estuary scale by using a combination of modeling approaches, existing literature, and expert opinion. Here, we identified thresholds in relationships between key metrics of fish communities (specifically fish species richness and harvestable fish abundance) with the extent of each of three ecosystems in estuaries (seagrass, mangroves, and oyster reefs) using generalized additive models (GAMs). These patterns were then interpreted based on our existing knowledge of the ecology and impacts facing each ecosystem from existing literature in the region, and expert opinion. Finally, we use spatially explicit Bayesian belief networks (BBNs) within individual estuaries to identify locations where restoring particular habitats will both be feasible and deliver the greatest ecological benefits for fish communities. This means that we modeled one restoration scenario for each ecosystem in each individual estuary. Here, each scenario is optimized based on our region-wide understanding of the effects of habitat extent on key features of fish assemblages, and is then applied to individual estuaries to ensure optimal outcomes. These scenarios address our study objectives because they incorporate a quantitative understanding of the effects of habitat extent on fish assemblages and apply these quantitative goals explicitly at two spatial scales across the region.

The Social, Economic, and Political Context of Restoration

The first five stages of systematic conservation planning principally deal with the social, economic, and political context of conservation efforts (Fig. 1). SEQ supports a human population of approximately 3.6 million people over a catchment of approximately 22,000 km^2 (Fig. 2). Extensive coastal urbanization in the region has resulted in substantial coastal habitat loss, and there is evidence to suggest that these losses have affected coastal fisheries (Gilby et al. 2019*b*). Lower fish abundance has socioeconomic effects by lowering the value of recreational and commercial fisheries. Therefore, the broader socioeconomic context for estuarine restoration in the region pivots mainly on coastal urbanization and fisheries. There is growing social and political will to restore key fish habitats, ideally investing in regional priority areas using best practice (Chenoweth EPLA and Bushland Restoration Services 2012). While political and financial support for some future restoration is encouraging and fulfills several of the first five steps in the framework, there exists only limited and fragmentary information to make any such investments effective and efficient in the region. The conservation goal for this case study is therefore to restore coastal ecosystems for the propose of returning the natural seascape to enhance their value for fish and fisheries.

Collecting Data on Biodiversity and Natural Features

Regional maps of marine habitats, land use, and water quality data are available from government agencies (Table S1). Land use maps and aerial imagery is accurate and regularly collected in the region, meaning that urbanized areas that would prohibit estuarine restoration from occurring are well known. Water quality data are sampled regularly (at least every 2 months) in each estuary (EHMP 2020), and so can be used to filter estuaries where water clarity and sedimentation would restrict the growth of seagrass and/or oyster reefs.

We obtained estuary-scale data on fish abundance and species richness using 10 × 1 hour deployments baited remote underwater video stations (BRUVS) distributed evenly from the

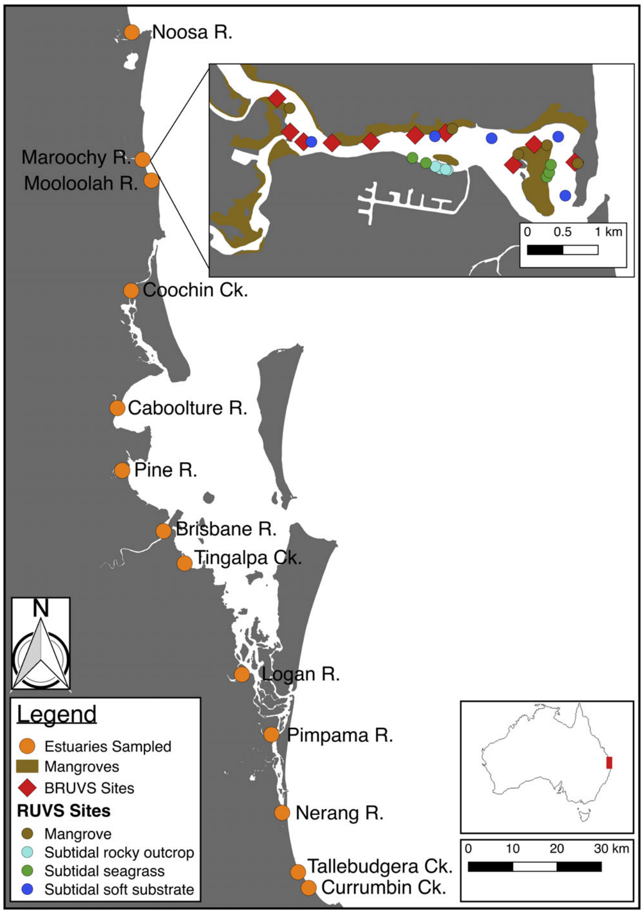

Figure 2. Sampled estuaries in southeast Queensland, Australia, with inset showing an example spread of sampling sites throughout each estuary.

estuary mouth to the point in each estuary where long-term (>10 year) winter salinity averaged >30 psu of each estuary (see Gilby et al. 2017) (Fig. 2). BRUVS consist of 3 kg weight that serves as a base and attachment point for cameras (GoPro recording at 1080p) and a polyvinyl chloride pole that holds the bait bag at 50 cm in front of the camera. The bait was 500 g of pilchards *Sardinops sagax* placed in a 20 × 30 cm mesh bag with 0.5 cm openings. BRUVS are preferred for quantifying these effects as they give a broader idea of general fish patterns within an area due to the aggregating of fish toward baits, as opposed to any habitat-specific effects. The value and optimal positioning of estuarine habitats to maximize fish abundance and species richness were surveyed using remote underwater video stations (RUVS; i.e. unbaited BRUVS) (see Gilby et al. 2018*b*) (Fig. 2). RUVS consist of 3 kg weight that serves as a base and attachment point for cameras (GoPro recording at 1080p). RUVS do not attract fish using baits, thereby avoiding the confounding effects of baited cameras drawing fishes from other habitats, and so are used to quantify fish-habitat associations (Gilby et al. 2018*b*). Fish assemblage composition was quantified from all videos using the standard *MaxN* statistic; the maximum number of individuals of each species identified in any single frame of each video. We calculated three key indicators of fish assemblages from each video: species richness being the number of unique species identified from each camera deployment, harvestable fish abundance being the sum of *MaxN* values for all species harvested commercially or recreationally in SEQ, and total fish abundance being the sum of *MaxN* values for all species identified from each camera deployment.

Setting Conservation Objectives

Historical data regarding the extent and condition of habitats is severely lacking, and significant changes to coastal hydrology and catchments due to urbanization have fundamentally modified the seascape in this region. Therefore, this study will focus on functional ecosystem extents in our case study, as opposed to any historical extent.

The diversity and abundance of fish are key social, economic, and cultural assets for local communities in SEQ. Thus, restoring fish habitats to maximize fish and fisheries is the prime restoration objective in this study. In the region, the value of an estuary for fish and fisheries is enhanced by seascapes that comprise a mosaic of alternative habitats available to fish, particularly the presence of healthy seagrass, mangroves, and oyster reefs (Gilby et al. 2018*b*). The quantitative restoration objective for this study is to restore the habitat matrix of seagrass, mangroves, and oyster reefs to *increase* total fish abundance and harvestable fish abundance *across the entire region*. Therefore, we model a restoration objective in which seagrass, mangrove, and oyster reefs are restored to estuaries such that fish abundance and diversity will be higher postrestoration than the 75th percentile fish abundance and diversity values from across the entire region during the baseline surveys. These 75th percentile values were calculated at the regional scale but were then applied separately to individual estuaries to calculate the extent of each habitat required to improve overall fish assemblage condition. Here, we chose 75th percentiles because they are a quantitative objective that is larger than both the mean and median values. In this sense, habitat restoration aims to raise the condition of marine habitats in all estuaries across the region to support overall higher fish community condition across the region.

Review Current Achievement of Objectives

Ecological restoration is employed in estuaries in the region, but greater expanses of restoration are required to meaningfully enhance fisheries (Saunders et al. 2017; Gilby et al. 2019*a*). For example, several oyster reef restoration projects in the region have demonstrated the efficacy of restoring oysters via resupplying settlement substrates when turbidity levels are low and spat supply is high, but these projects remain relatively small (<100s of meters) (Diggles et al. 2019; Gilby et al. 2019*a*). These considerations regarding the success of current projects are incorporated into the local-scale prioritization of habitat placement (see Identify Potential Restoration Sites Within Planning Units section).

Target Actions for Planning Units

Once quantitative and measurable objectives are in place, these form the basis from which to start a systematic and quantitative evaluation about the choice of methods, sites, and connectivity (Fig. 3A). Where restoration is the most effective action, the next step is to rank the allocation of restoration investment to regional planning units, contrasting the extent of existing habitat patches in each unit against the stated restoration objectives.

We prioritized restoration at two spatially nested scales: (1) at the regional scale, we used individual estuaries as planning units as this represents the scale at which most management actions are currently implemented in our study region (EHMP 2020), and (2) at the local scale (i.e. within estuaries) we used 10×10 m grid cells of the planning unit, because this is the scale at which many local restoration plots are often selected, and this exceeds the resolution of local habitat maps.

We first calculated the current extent of estuarine habitat types and corrected these values for the size of the estuary (see Table S1). The second step was to use statistical models and the quantitative objective to determine actions for each planning unit and set their relative priority at a regional scale (Fig. 3). We used GAMs in the *mgcv* package (Wood 2019) of R to calculate relationships between fish species richness and harvestable fish abundance (response variables, extracted from the broad-scale baited underwater videos described earlier) and the extent of the three habitats (i.e. mangroves, seagrass, and oyster reef) in individual estuaries. Models included the extent of mangroves, seagrass, and oyster reefs, so relationships for each habitat account for the effects of the other two habitats. GAM overfitting was limited by restricting models to four knots or fewer ($k = 4$). We then used these models along with the current understanding of the ecology and extent of ecosystems within the

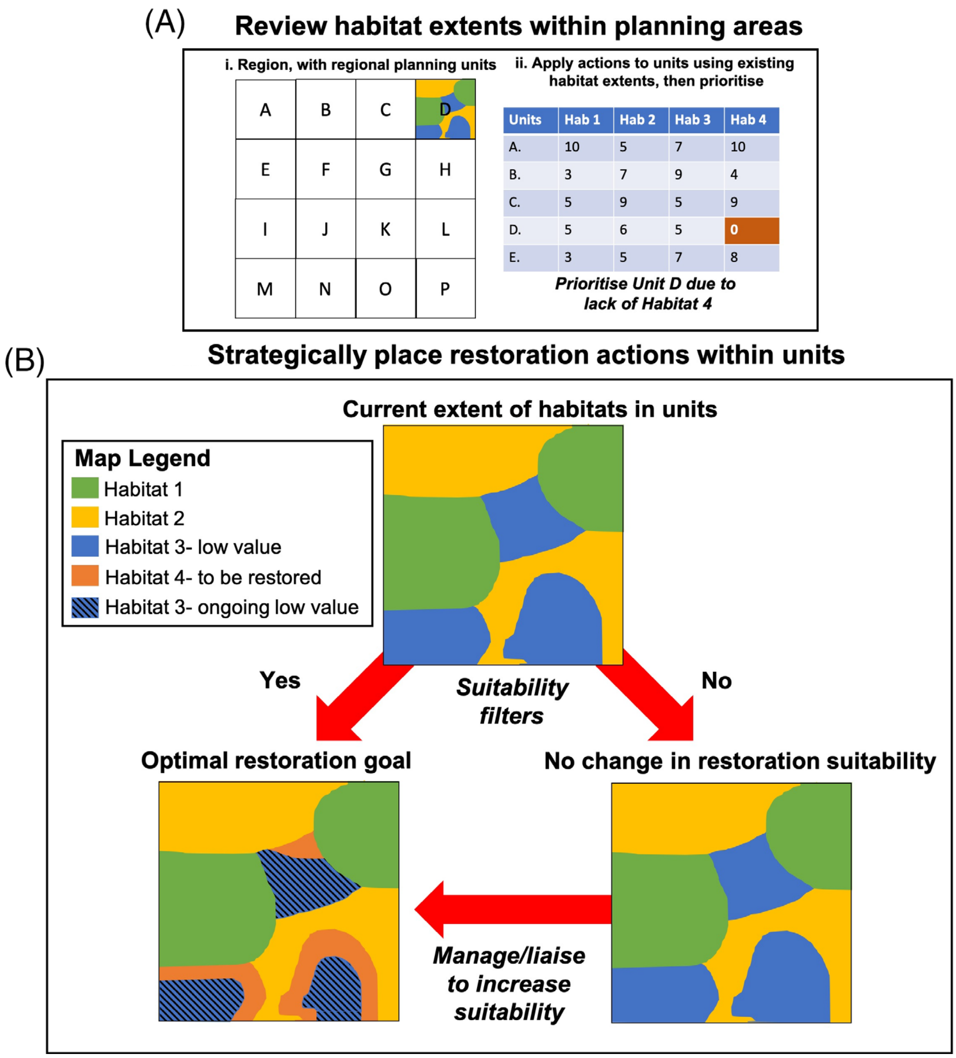

Figure 3. Prioritizing habitat restoration at two spatial scales using data-driven systematic landscape restoration. (A) We used preestablished regional planning units (i.e. A–P in Panel i), and the extents of habitats within those planning units (Panel ii) to prioritize actions for each regional planning unit, and the order in which these actions should be applied. (B) We then prioritize restoration site placement within planning unit D to restore lost habitats (Habitat 4). Where suitability is low, then additional management interventions are required before commencing habitat restoration.

Table 1. Average (±standard error [SE]) of current-day species richness and harvestable fish abundance, and average long-term turbidity across all estuaries surveyed in this study.

Estuary	*Species Richness*	*SE*	*Harvestable Fish*	*SE*	*Turbidity (Nephelometric Turbidity Units)*
Brisbane River	2.85	0.44	2.75	0.70	58.1
Caboolture River	4.90	0.32	8.55	2.14	17.5
Coochin Creek	2.65	0.31	6.98	1.17	13
Currumbin Creek	3.50	0.37	44.23	26.74	8.1
Logan River	2.65	0.31	2.40	0.83	74.7
Maroochy River	3.35	0.45	16.13	2.72	14.7
Mooloolah River	3.25	0.50	9.50	1.59	14.6
Nerang River	5.40	0.60	27.73	7.71	8.2
Noosa River	3.15	0.43	13.58	1.84	17.7
North Pine River	4.95	0.53	9.58	2.75	10
Pimpama River	3.35	0.26	12.05	2.57	16.6
Tallebudgera Creek	4.25	0.38	22.95	3.82	5.3
Tingalpa Creek	5.75	0.47	20.98	3.55	12.3

region from both published articles and expert opinion to calculate the habitat area at which fish species richness and abundance reached the target 75th percentiles. These 75th percentile habitat extent values were the targets for habitat restoration at the regional scale, and were used to calculate the extent of each habitat that is required to be restored to each estuary. Therefore, we set extent goals using a percentile values across the entire region, but calculated individual extents required for individual estuaries.

We created a categorical approach that scales the relative priority of each estuary for restoration actions based on our understanding that it is the mix of mangroves, seagrass, and oyster reefs that contribute most to fish diversity and abundance within this region (Gilby et al. 2018*b*):

- *Very high* restoration priority estuaries are those where one or more habitats has become extirpated;
- *High* restoration priority estuaries support all three habitat types, but the extent of mangroves *or* seagrass is below the threshold for species richness *and* harvestable fish abundance;
- *Medium* restoration priority estuaries support all three habitats, but the extent of mangroves *or* seagrass is below the threshold for species richness *or* harvestable fish abundance (i.e. one of the fish targets is met);
- *Low* restoration priority estuaries come in two forms:
  - estuaries in which mangroves and seagrass are above the threshold for *both* species richness *and* harvestable fish abundance, or
  - estuaries with high turbidity or urbanization that preclude restoration, especially where reestablishing oyster or restoring seagrass is required.

Identify Potential Restoration Sites Within Planning Units

After identifying priority restoration actions at regional scales, the next stage identifies restoration sites at narrower spatial scales within regional planning units (Fig. 3). To do this, we used BBNs in Netica v5.12 (Norsys Software 1997) with the resulting outputs then mapped (using 10 × 10 m pixels) in QGIS to create spatially explicit BBNs that reflect restoration suitability within individual estuaries. We compiled separate spatially explicit BBNs for seagrass, mangroves, and oyster reefs because these habitats require fundamentally different environmental conditions, and so have different BBN nodes and relationships (Fig. S1). Relationships between BBN nodes were entered manually using published ecological information for each of the three habitats being restored (Tables S1–S3, Fig. S1). We accounted for the likely benefits of restoration actions for fish within each 10 × 10 m pixel by incorporating layers that relate to the effects of ecological connectivity for fish. We accounted for the likely feasibility of restoration actions within each pixel by incorporating layers that relate to the growing conditions required for each habitat, and existing estuary uses by incorporating layers of current uses that might limit restoration actions (e.g. the presence of aquaculture, ports, or mooring areas). Full details on the input nodes (Table S2), relationships between nodes (Fig. S1), and conditional probability tables (Table S3) for each BBN are available as supporting information.

Results

Target Actions for Each Regional Planning Unit

GAMs showed significant effects of both mangrove (Fig. S2A) and seagrass (Fig. S2B) extent on fish species richness and harvestable fish abundance. Regional threshold values to be met in habitat restoration were five species for species richness and 17 individuals for harvestable fish abundance (Table 1); consequently, these values represent the target attributes of fish assemblages in each estuary once restoration is complete. Results of the GAMs were complex, and so required significant consideration prior to setting regional extent goals.

We found that species richness and harvestable fish abundance had nonlinear relationships with the extent of mangroves (Fig. S2A). We set the regional extent goal for mangroves at 6 m^2/m because this extent maximized harvestable fish abundance, and minimized reductions in species richness with very extensive mangrove forests. This was a particularly challenging decision because maximizing mangrove extent at 22.1 m^2/m (i.e. the target extent for harvestable fish abundance) would have a negative consequence for species richness within estuaries, likely due to the homogenizing effects of extensive mangrove forests in estuarine seascapes (Henderson et al. 2019).

Extent goals for seagrass were the most straightforward to set. Estuaries within the study region require at least 1% of the estuary to be covered in seagrass to maximize both species richness and harvestable fish abundance (Fig. S2B). Consequently, 1% is the regional extent goal for seagrass (Table 2).

We found no significant relationship between oyster reef extent and fish (Fig. S2C), but several estuaries had no remaining oyster reefs and reefs were small, isolated, and scattered fragments in others. The maximum extent of oyster reefs present in estuaries in this region was 0.579%; this is likely to be a low estimate of the target that restoration should reach for each estuary given the history of oyster reef degradation in the region. Despite this, and for the purposes of this study, we set the oyster reef target extent to this maximum extent of 0.579%. Consequently, this value represents a conservatively low restoration goal for oyster reefs in this region. Further refinement of the extent goal could be achieved through historical studies and/or stakeholder engagement activities.

All regional planning units were assessed as requiring some restoration according to our quantitative goals. Five estuaries were assessed to have *very high* restoration priority, three as *high*, two as *medium*, and three as *low* (Table 2). Two estuaries would require significantly improved water column turbidity through catchment management before in-water habitat restoration could be recommended (Brisbane and Logan rivers; Table 1), and two are so severely urbanized that space for restoration would be limited. Two estuaries required actions on only one ecosystem, nine estuaries required action to restore two habitats, and two required restoration of all three habitat types (Table 2). Estuaries identified as "low" priority based on high turbidity or extensive urbanization would also require habitat restoration.

Identify Potential Restoration Sites Within Planning Units

We chose the Mooloolah River estuary to illustrate how the spatial allocation of restoration investments can be optimized at the local level. Seagrass is not currently recorded in the Mooloolah River, has likely become extirpated and hence requires restoration (Table 2). Site selection for seagrass restoration might conflict with other potential restoration actions in this estuary (i.e. replanting of mangroves; Table 2).

Areas of "good restorability" for seagrass are throughout the middle stretch of the estuary where there are current stands of mangroves (Fig. 4A). The spatial selection for seagrass was driven by the proximity of sites to mangroves, as previous studies have established that seagrass sites contain more diverse and abundant fish assemblages (i.e. our measure of restoration "benefit") when they are positioned more closely than this threshold distance to mangroves (Gilby et al. 2018*b*). Preferred sites for mangrove restoration were, therefore, also predominantly positioned in the central stretch of the estuary where mangroves are (Fig. 4B). There was, however, little conflict between "good restorability" sites for seagrass and mangroves because mangroves and seagrasses grow at different water depths (Fig. 4).

Yellow, "low restorability" areas for both seagrass and mangroves were mostly toward the mouth of the estuary because there are no mangroves currently in this section of the estuary (Fig. 4). Mangroves would be relatively easy to restore in these parts where intertidal areas are sedimentary shores and there is natural recruitment of mangroves. However, these new mangroves would be poorly connected to existing forests and hence their ecological value as fish habitats is predicted to be low. Moreover, there are some social costs as there is extensive dredging (which would physically damage restoration sites) and canal estates (where people do not want mangroves to grow) toward the entrance to the Mooloolah River which significantly reduces restoration feasibility.

Discussion

The extent and scale of habitat restoration is increasing globally (Shoo et al. 2017; Tobon et al. 2017; Gilby et al. 2018a). It is, therefore, important to strategically place restoration efforts across landscapes to maximize their ecological benefits while reducing any impacts on social, cultural, or economic values connected to landscapes (Rappaport et al. 2015). In this study, we applied systematic conservation planning to address the challenge of prioritizing restoration actions at multiple spatial scales; a challenge perhaps unique to restoration given the narrow spatial scale required to allocate restoration within planning units to ensure success (i.e. <10s m scale). Here, restoration is spatially nested and sequential, and is prioritized at a regional scale to achieve landscape scale, regional goals for restoration, and then placed at a local scale within planning units to maximize potential benefits by considering site connectivity and landscape context. Optimizing restoration outcomes by understanding the relative importance of habitats for reaching biodiversity goals is vital when prioritizing landscape restoration actions (Cattarino et al. 2016). Applying a systematic planning framework to restoration provides multiple opportunities to incorporate these data and ensures that quantitative objectives are set for the restoration actions, an ongoing challenge for restoration (Guerrero et al. 2017; Tobon et al. 2017).

A systematic approach for prioritizing restoration effort will assist in maximizing outcomes in several ways. On-the-ground restoration remains evenly spread along the continuum from top-down (mostly regional government actions) to bottom-up (mostly community-based actions) approaches (Ban et al. 2011). Having a broad, regional scale and structured approach followed by a focused, local context means that broader restoration goals can be prioritized by regional management bodies, and then focused at the local scale via the allocation of funding or effort. As the extent of restoration projects increases, the likelihood of disruptions to existing uses increases, there are fewer optimum restoration sites to select from, and the amount of financial investment increases (Miller & Hobbs 2007; Bullock et al. 2011). Therefore, prioritizing the restoration of habitats according to (1) their importance for achieving stated restoration goals, (2) the feasibility of

Table 2. Estuaries of southeast Queensland ranked according to habitat restoration needs for mangroves, seagrass, and oyster reefs, and the extent of each habitat required to reach goals. Goals for the extent of each ecosystem are based on the 75th percentile values of species richness of fish and the abundance of harvestable fish across the entire region. Criteria denotes habitat extent that currently fails the benchmark for fish species richness[(s)], fish abundance[(a)], or both[(s,a)]. Oyster reefs have been massively and widely reduced in all estuaries, with only small fragments remaining at most. Therefore, all estuaries require restoration of reefs to at least the best remaining cover. Bold values denote current habitat sizes that fail the target for fish. Required areas are in addition to existing extents within each estuary and represent the extent of each habitat required (in ha) to meet the restoration goals.

	Mangroves				*Seagrass (%)*				*Oyster Reefs (%)*			
Restoration Priority and Estuary	*Current extent (m^2/m)*	*Criteria*	*Required %*	*Required (ha)*	*Current extent (%)*	*Criteria*	*Required %*	*Required (ha)*	*Current extent (%)*	*Criteria*	*Required %*	*Required (ha)*
Very high												
Mooloolah River	6		0	0	**0**	s,a	1	0.9	**0.14**	s,a	0.439	0.39
Currumbin Creek	6.1		0	0	**0.874**	s	0.126	0.1	**0**	s,a	0.579	0.24
Caboolture River	32.7		0	0	**0.106**	s,a	0.894	4.4	**0**	s,a	0.579	2.84
North Pine River	45.5		0	0	**0**	s,a	1	4.3	**0.108**	s,a	0.471	2.01
Coochin Creek	23		0	0	**0**	s,a	1	1.0	**0.579**	s,a	0	0
High												
Noosa River	12.5		0	0	**0.148**	s,a	0.852	9.9	**0.006**	s,a	0.573	6.69
Tallebudgera Creek	**5.1**	s,a	0.9	0.4	**0.343**	s	0.657	0.6	**0.416**	s,a	0.163	0.16
Maroochy River	24		0	0	**0.165**	s,a	0.835	5.2	**0.023**	s,a	0.556	3.46
Medium												
Tingalpa Creek	21.8		0	0	1.017		0	0	**0.103**	s,a	0.476	0.81
Pimpama River	46.9		0	0	**0.839**	s	0.161	0.3	**0.067**	s,a	0.512	0.82
Low												
Logan River	34.3		0	0	**0**	s,a	1	9.4	**0.004**	s,a	0.575	5.38
Brisbane River	11.2		0	0.0	**0**	s,a	1	31.5	**0**	s,a	0.579	18.23
Nerang River	**0**	s,a	6	12.8	**0.516**	s	0.484	1.5	**0.002**	s,a	0.577	1.81
Restoration goals (75th percentiles)												
Habitat extent required for 75th percentile species richness	6				1				0.579			
Habitat extent required for 75th percentile harvestable fish	22.1				0.27				0.579			

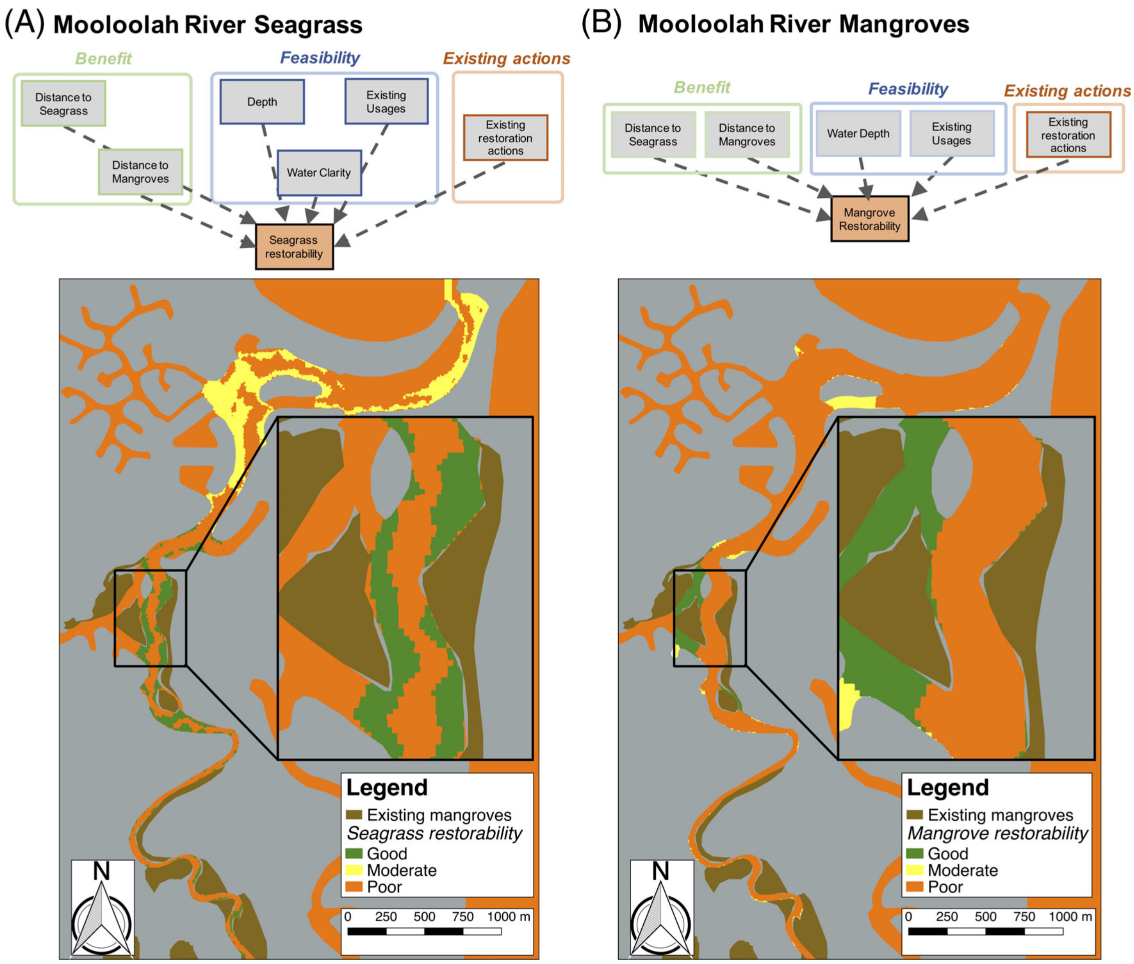

Figure 4. Model frameworks and outputs of spatially explicit Bayesian belief networks illustrating potential restoration sites for: (A) seagrass, a habitat type that current regional mapping data denote as locally not recorded at the scale of available mapping; and (B) mangroves, a habitat currently below the regional region median extent, in the Mooloolah River, eastern Australia.

restoration in the area, and (3) the likely cost per unit area becomes increasingly important. While it is clear that concepts from landscape and spatial ecology can be used to select restoration sites and maximize outcomes for biodiversity, few projects actually implement these concepts in restoration plans (Langhans et al. 2016; Gilby et al. 2018a). Systematic and spatially explicit approaches like the one used here can incorporate data on the distribution of species and habitats to identify potential restoration sites where environmental conditions are appropriate, and that provide the greatest ecological benefit.

At the scale of local planning units, there can be conflicts regarding which habitats to restore in which locations because different restoration actions might compete for available space. Indeed, it is likely in most scenarios that multiple ecosystems could be restored across a landscape. In estuaries in SEQ, it is understood that it is the mix of ecosystems present within individual estuaries that drives fish abundance and diversity, as opposed to homogeneous swathes of individual ecosystems (Gilby et al. 2018*b*; Henderson et al. 2019). Therefore, in our case study, restoration was prioritized for ecosystems that were completely missing from planning units. In situations where a clear ranking of actions is not possible, a balance must be struck between restoring different habitat types according to the likelihood of successful restoration and potential benefits (Adame et al. 2015). An alternative approach is to identify estuaries which have the full mix of habitats but require only a small amount of restoration to bring them to the quantitative target, thereby bringing many planning units up to "fully restored" status. In essence, the decision here is made initially at the regional planning unit scale, and then narrowed down to one habitat or the other within regional planning units based on the potential benefits of restoration and the total areas of "good" restorability areas available for each habitat (Wilson et al. 2011). Historical information regarding the previous cover and distribution of ecosystems within a region might be useful as a guide in these situations (Balaguer et al. 2014), noting that using historical baselines as a rule, as opposed to a guide, is often inappropriate due to the effects of extensive urbanization, species invasions, climate change, or other exogenic threats that are too great (Thorpe & Stanley 2011). These potential differences in prioritization approaches highlight the need to engage with stakeholders to establish their desired prioritization approach, and account for available budgets in establishing regional prioritizations.

Maximizing cost effectiveness is an important consideration in conservation planning (Laycock et al. 2009). Cost effectiveness encompasses the likelihood of restoration success (feasibility), the likely benefits of restoring the required amount of habitat (benefit), and the total costs (principally financial, but often also social or political) of reaching conservation goals (Klein et al. 2017). These components of cost effectiveness can be included in spatial modeling approaches, as conducted here through the inclusion of key spatial layers. For example, in this study, we included layers of existing human land uses in BBNs that precluded restoration from occurring in these areas. Future iterations of these models could include maps depicting variability in costs for restoring in different parts of estuaries (due to, e.g. accessibility), social costs (e.g. reductions in access for recreational boating), and/or variability in restoration costs due to existing habitat condition (e.g. some sites may need remediation prior to restoration); indeed the lack of these sorts of considerations is a weakness of the approach we undertook here, and need to be strengthened to enable a proper analysis of costs versus benefits. Alternatively, they can be used to quantify cost effectiveness of actions within individual planning units, and then prioritized formally in order of cost effectiveness. Similarly, using coordinated efforts to restore ecosystems across landscapes has several key benefits. First, the combined effects of individual restoration actions that are coordinated at broader spatial scales result in social and economic benefits and costs being more thoroughly considered (Roy et al. 2018). Second, coordinating approaches more broadly results in greater efficiency in the allocation of efforts and funds to reach these heightened social and ecological benefits (Neeson et al. 2015).

After targeting actions for planning units, and selecting sites within planning units, the next stage of systematic landscape restoration applies the actions on the ground. Habitats should be restored according to international best practice, and to reflect the condition and structure (e.g. plant density and composition, structural complexity, geometry) of extant high-quality habitat in the region (i.e. target or reference ecosystem conditions) (Gann et al. 2019). Stages following implementation involve managing threats beyond the footprint of restored habitats, using data from monitoring programs to determine if goals are being reached, and implementing changes where necessary. Restored habitats should, therefore, be monitored to evaluate effectiveness (Gilby et al. 2018*c*), and restoration plans should be modified, if necessary, to improve performance against restoration goals.

We modeled scenarios that targeted restoration of habitats to the extent that supports the 75th percentile of fish species richness and abundance across the region, and applied these region-wide learnings to actions for individual estuaries. This value was chosen because it served to increase the overall carrying capacity of the region for harvestable fish abundance and maintain biodiversity more evenly across estuaries at a higher level than simply the region mean or median. In this sense, we are confident that restoring to the extents we targeted for each ecosystem would result in an increase in the abundance and diversity of fish assemblages across the region, thereby meeting the conservation objective of enhancing fish assemblages. We acknowledge, though, that this value could potentially be modified based on stakeholder consultation. For example, a community may seek to restore only to the regional mean or median (i.e. 50th percentile) values for fish assemblages or may seek to be more adventurous and raise fish communities to the 90th percentile. Under such a scenario, more restoration would obviously be required, so this decision may also be made on the basis of available funds.

Further stakeholder consultation would certainly be required for optimizing mangrove restoration extents in this study given the complexities that we identified in relationships between habitat extent and fish assemblages. Here, we identified a decline in harvestable fish abundance after a threshold of approximately 10 m/m^2 of mangroves in the estuaries sampled, and rapid declines in species richness with increasing mangrove extent. These results are somewhat unsurprising, as previous studies in the region have shown that fish abundance and diversity can decline at higher mangrove extents because extensive mangrove forests may homogenize seascapes (Henderson et al. 2019). In this sense, stakeholders would need to be briefed and engaged on the decision of setting the regional extent goal at 6 m^2/m versus the more adventurous 22.1 m^2/m that maximizes both habitat extent and harvestable fish abundance, despite the fact that this goal would potentially not meet the overarching goal of restoring to the 75th percentile for all ecosystems and estuaries. An alternative option would have been to set a range as the extent goal for mangroves, with the minimum extent being zero and the maximum extent being 6 m^2/m, which maximized both species richness and harvestable fish abundance. We chose against this strategy, however, for two key reasons. First, setting an extent goal of zero sends the wrong message to stakeholders and the public engaging in restoration planning (i.e. that mangroves are an unnecessary ecosystem, and you could have zero without any problems). Second, it is well established that fish assemblages in the region rely upon the matrix of all estuarine habitats, and this is therefore reflected in the overall restoration objective "to restore the habitat matrix" across estuaries. Consequently, setting an extent range with a minimum of zero fails overall the restoration objective.

We modeled restoration priorities based on the likely benefits for fish and fisheries, but we acknowledge that there might be other motivations for habitat restoration in these systems. These benefits are not always ecological; for example, some coastal restoration prioritizes for shoreline stabilization, nutrient sequestration, flood mitigation, or recreational and aesthetic benefits. Identifying "bright spots" for restoration that maximize outcomes for multiple restoration goals would, therefore, be a useful extension for the framework we have presented here (Gilby et al. 2019*a*). BBNs could be made more comprehensive by incorporating additional datasets (e.g. habitat quality, hydrology, genetic connectivity/viability; not yet available in this region) to inform spatial prioritization exercises. In our case study, we only analyzed direct restoration of marine habitats. A broader, cross-realm approach to restoration that also considered catchment revegetation could consider actions to improve water quality when prioritizing restoration efforts (Saunders et al. 2017).

In the face of continuing global habitat loss and degradation, it is crucial that sound and defensible decisions are made about which habitats to restore and where to restore them. These decisions typically involve a trade-off between improving ecological values, minimizing any negative consequences for users (e.g. recreational and aesthetic considerations), and minimizing financial costs. It is possible to optimize the placement of restoration investments using quantitative goals and empirical data on habitat values and restoration feasibility; indeed, these approaches should become the norm. The challenge before us is to use these approaches to design and implement better planning techniques that systematically prioritize restoration investments to achieve better restoration outcomes at lower costs.

Acknowledgments

This work was funded by Noosa Biosphere Reserve Foundation, Noosa Council, Healthy Land and Water, and SeaWorld Research and Rescue Foundation grants to B.L.G., A.D.O., C.J.H., and T.A.S. The authors thank N. Yabsley, H. Borland, S. Thackwray, B. Gorrissen, B. Frohloff, and C. Duncan for assistance with field work and video analyses. C.J.B. was supported by a Discovery Early Career Researcher Award (DE160101207) from the Australian Research Council. C.J.B. and R.M.C. were supported by The Global Wetlands Project and a Discovery Project from the Australian Research Council (DP180103124).

LITERATURE CITED

Adame MF, Hermoso V, Perhans K, Lovelock CE, Herrera-Silveira JA (2015) Selecting cost-effective areas for restoration of ecosystem services. Conservation Biology 29:493–502

Balaguer L, Escudero A, Martin-Duque JF, Mola I, Aronson J (2014) The historical reference in restoration ecology: re-defining a cornerstone concept. Biological Conservation 176:12–20

Ban NC, Adams VM, Almany GR, Ban S, Cinner JE, Mccook LJ, Mills M, Pressey RL, White A (2011) Designing, implementing and managing marine protected areas: emerging trends and opportunities for coral reef nations. Journal of Experimental Marine Biology and Ecology 408:21–31

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. Ecological Applications 26:1055–1074

Bullock JM, Aronson J, Newton AC, Pywell RF, Rey-Benayas JM (2011) Restoration of ecosystem services and biodiversity: conflicts and opportunities. Trends in Ecology & Evolution 26:541–549

Cattarino L, Hermoso V, Bradford LW, Carwardine J, Wilson KA, Kennard MJ, Linke S (2016) Accounting for continuous species' responses to management effort enhances cost-effectiveness of conservation decisions. Biological Conservation 197:116–123

Cattarino L, Hermoso V, Carwardine J, Adams VM, Kennard MJ, Linke S (2018) Information uncertainty influences conservation outcomes when prioritizing multi-action management efforts. Journal of Applied Ecology 55: 2171–2180

Chenoweth EPLA and Bushland Restoration Services (2012) *South East Queensland ecological restoration framework: code of practice*. Prepared on behalf of SEQ Catchments and South East Queensland Local Governments, Brisbane, Australia

Diggles BK, Porter R, Vardon J, Hawthorne S, Veary E (2019) Pumicestone Passage Shellfish Reef Habitat Restoration Project—December 2018 deployment. 6 Month invertebrate monitoring. OzFish, Brisbane, Australia

EHMP (Ecosystem health monitoring program) (2020) https://hlw.org.au/project/ecosystem-health-monitoring-program/ (accessed 7 Apr 2018)

Gann GD, McDonald T, Walder B, Aronson J, Nelson CR, Jonson J, et al. (2019) International principles and standards for the practice of ecological restoration. Second Edition. Restoration Ecology 27:S1–S46

Gilby BL, Olds AD, Connolly RM, Yabsley NA, Maxwell PS, Tibbetts IR, Schoeman DS, Schlacher TA (2017) Umbrellas can work under water: using threatened species as indicator and management surrogates can improve coastal conservation. Estuarine, Coastal and Shelf Science 199:132–140

Gilby BL, Olds AD, Connolly RM, Henderson CJ, Schlacher TA (2018a) Spatial restoration ecology: placing restoration in a landscape context. Bioscience 68:1007–1019

Gilby BL, Olds AD, Connolly RM, Maxwell PS, Henderson CJ, Schlacher TA (2018b) Seagrass meadows shape fish assemblages across estuarine seascapes. Marine Ecology Progress Series 588:179–189

Gilby BL, Olds AD, Peterson CH, Connolly RM, Voss CM, Bishop MJ, Elliott M, Grabowski JH, Ortodossi NL, Schlacher TA (2018c) Maximising the benefits of oyster reef restoration for finfish and their fisheries. Fish and Fisheries 19:931–947

Gilby BL, Olds AD, Duncan CK, Ortodossi NL, Henderson CJ, Schlacher TA (2019a) Identifying restoration hotspots that deliver multiple ecological benefits. Restoration Ecology 28:222–232

Gilby BL, Olds AD, Henderson CJ, Ortodossi NL, Connolly RM, Schlacher TA (2019b) Seascape context modifies fish respond to restored oyster reef structures. ICES Journal of Marine Science 76:1131–1139

Guerrero AM, Shoo L, Iacona G, Standish RJ, Catterall CP, Rumpff L, De Bie K, White Z, Matzek V, Wilson KA (2017) Using structured decision-making to set restoration objectives when multiple values and preferences exist. Restoration Ecology 25:858–865

Hallett LM, Diver S, Eitzel MV, Olson JJ, Ramage BS, Sardinas H, Statman-Weil Z, Suding KN (2013) Do we practice what we preach? Goal setting for ecological restoration. Restoration Ecology 21:312–319

Henderson CJ, Gilby BL, Schlacher TA, Connolly RM, Sheaves M, Maxwell PS, et al. (2019) Landscape transformation alters functional diversity in coastal seascapes. Ecography 43:138–148

Hermoso V, Clavero M, Filipe AF (2021) An accessible optimisation method for barrier removal planning in stream networks. Science of the Total Environment 752:141943

Higgs E, Falk DA, Guerrini A, Hall M, Harris J, Hobbs RJ, Jackson ST, Rhemtulla JM, Throop W (2014) The changing role of history in restoration ecology. Frontiers in Ecology and the Environment 12:499–506

Klein CJ, Beher J, Chaloupka M, Hamann M, Limpus C, Possingham HP (2017) Prioritization of marine turtle management projects: a protocol that accounts for threats to different life history stages. Conservation Letters 10:547–554

Langhans SD, Gessner J, Hermoso V, Wolter C (2016) Coupling systematic planning and expert judgement enhances the efficiency of river restoration. Science of the Total Environment 560-561:266–273

Laycock H, Moran D, Smart J, Raffaelli D, White P (2009) Evaluating the cost-effectiveness of conservation: the UK biodiversity action plan. Biological Conservation 142:3120–3127

Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405: 243–253

Miller JR, Hobbs RJ (2007) Habitat restoration—do we know what we're doing? Restoration Ecology 15:382–390

Neeson TM, Ferris MC, Diebel MW, Doran PJ, O'hanley JR, Mcintyre PB (2015) Enhancing ecosystem restoration efficiency through spatial and temporal coordination. Proceedings of the National Academy of Sciences 112:6236–6241

Norsys Software (1997) Netica user's guide. Norsys Software, Vancouver, Canada

Possingham H, Ball I, Andelman S (2000) Mathematical methods for identifying representative reserve netorks. Pages 291–305.In: Ferson S, Burgman M (eds) Quantitative methods for conservation biology. Springer-Verlag, New York

Pressey RL, Bottrill MC (2009) Approaches to landscape- and seascape-scale conservation planning: convergence, contrasts and challenges. Oryx 43:464

Rappaport DI, Tambosi LR, Metzger JP, Cadotte M (2015) A landscape triage approach: combining spatial and temporal dynamics to prioritize restoration and conservation. Journal of Applied Ecology 52:590–601

Roy SG, Uchida E, De Souza SP, Blachly B, Fox E, Gardner K, et al. (2018) A multiscale approach to balance trade-offs among dam infrastructure, river restoration, and cost. Proceedings of the National Academy of Sciences 115:12069–12074

Saunders MI, Bode M, Atkinson S, Klein CJ, Metaxas A, Beher J, et al. (2017) Simple rules can guide whether land- or ocean-based conservation will best benefit marine ecosystems. PLoS Biology 15:e2001886

Shoo LP, Catterall CP, Nicol S, Christian R, Rhodes J, Atkinson P, Butler D, Zhu R, Wilson KA (2017) Navigating complex decisions in restoration investment. Conservation Letters 10:748–756

Simenstad C, Reed D, Ford M (2006) When is restoration not? Incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. Ecological Engineering 26:27–39

Thompson BA (2011) Planning for implementation: landscape-level restoration planning in an agricultural setting. Restoration Ecology 19:5–13

Thorpe AS, Stanley AG (2011) Determining appropriate goals for restoration of imperilled communities and species. Journal of Applied Ecology 48: 275–279

Tobon W, Urquiza-Haas T, Koleff P, Schroter M, Ortega-Alvarez R, Campo J, Lindig-Cisneros R, Sarukhan J, Bonn A (2017) Restoration planning to guide Aichi targets in a megadiverse country. Conservation Biology 31: 1086–1097

Wilson KA, Lulow M, Burger J, Fang YC, Andersen C, Olson D, O'Connell M, Mcbride MF (2011) Optimal restoration: accounting for space, time and uncertainty. Journal of Applied Ecology 48:715–725

Wood S. (2019) mgcv: mixed GAM computation vehicle with GCV/AIC/REML smoothness estimation. R package version 1.8-24.

Supporting Information

The following information may be found in the online version of this article:

Figure S1. Bayesian belief network frameworks for (A) seagrass, (B) mangroves, and (C) oyster reef restoration in southeast Queensland, Australia.

Figure S2. Generalized additive model plots showing relationships between species richness (top line) and harvestable fish abundance (bottom line).

Table S1. List of estuary-scale factors, their definitions, and sources.

Table S2. List of nodes, their relationship with the restorability node, relationship justifications, data sources, and supporting references for spatially explicit Bayesian network analyses.

Table S3. Conditional probability tables for identifying (A) oyster reefs, (B) mangroves, and (C) seagrasses.

Coordinating Editor: Maksym Polyakov

Received: 5 November, 2020; First decision: 12 December, 2020; Revised: 18 March, 2021; Accepted: 28 March, 2021