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Coral reef metrics and habitat equivalency analysis

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A B S T R A C T

When coral reefs held in United States public trust are injured by incidents such as vessel groundings or oil spills, a natural resource damage assessment (NRDA) process may be conducted to quantify the resource service loss. Coral cover has been used as an indicator metric to represent lost services in habitat equivalency analyses for determination of compensatory restoration. Depending on the injury and habitat, however, lost services may be more comprehensively represented by alternative approaches such as composite metrics which incorporate other coral reef community characteristics, or a resource-scale approach utilizing size-frequency distributions of injured organisms. We describe the evolving state of practice for capturing coral reef ecosystem services within the natural resources damage assessment context, explore applications and limitations of current metrics, and suggest future directions that may increase the likelihood that NRDA metrics more fully address ecosystem services affected by an injury.

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1. Introduction

Coral reef injuries can be caused by human disturbances, including vessel groundings [1–4], anchor drops [5], towline abrasions [2], lost vessel cargo crushing benthic habitat, oil spills [6,7], dredging [4,7], and beach renourishments [4,8]. If an injury falls under specific U.S. legal statutes [9–12], restoration actions may be taken to restore lost ecological services. The ecological goal of the natural resource damage assessment (NRDA) process is to achieve successful functional replacement of lost services using metrics appropriate to both the injury and to ecological context of the habitat to quantify the restoration requirement. Before we describe how coral reef metrics are currently used in the NRDA process, we first review the existing NRDA legal and economic framework as applicable to coral reefs.

1.1. NRDA framework

In the United States and jurisdictional waters, the National Marine Sanctuaries Act (NMSA) [9] and the National Park System Resource Protection Act (NPSRPA) [12] provide location-specific authority for resource trustees (various federal and state governments) to recover damages for injuries to trust resources, such as coral reefs, under certain circumstances. States may be co-trustees with federal agencies in these otherwise federally-defined jurisdictions and may also have their own state-legislated provisions. For coral reef injuries outside of the Sanctuaries and Parks, U.S. trustees may seek damages under the Oil Pollution Act (OPA) when resources have been injured as a result of an oil spill or activities taken in order to alleviate an imminent threat of an oil spill [10], or the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) if the cause of injury is due to the release of a contaminant [11]. Though current law provides recourse when coral reefs are injured in selected circumstances and geographic locations, many coral reef injury events are not covered by these authorities. However, the ability of trustees to seek damages for coral reef injuries in U.S. waters would be greatly enhanced by the proposed Coral Reef Ecosystem Conservation Amendments Act that was delivered to the U.S. Congress by the Administration. Both chambers of the 110th Congress introduced legislation to reauthorize the Coral Reef Conservation Act of 2000. The Administration, House, and Senate versions of this legislation would provide, for the first time, the authority to pursue those liable for injury to all coral reefs in U.S. waters.

With a pending reauthorization of the Coral Reef Conservation Act, it is an opportune time to examine current coral reef NRDA approaches and suggest refinements for future applications within the economic, ecological, and legal constraints of NRDA. This paper is intended to serve as a bridge between coral reef restoration ecology and the economic and legal guidelines of NRDAs as applied to U.S. trust resources. We describe the evolving state of practice for capturing coral reef ecosystem services within the NRDA context, explore applications and limitations of current metrics, and suggest

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future directions that may increase the likelihood that coral reef damage assessments, restoration, and monitoring efforts more fully address the coral reef ecosystem services affected by an injury.

1.2. Valuation of lost services

Under U.S. statutes, the amount of damages for an injury to resources held in public trust is determined by an NRDA that combines legal, economic, and ecological processes. NRDAs are non-punitive in nature, and funds recovered from those responsible are used solely to pay for the trustees' costs associated with damage assessment and to restore, replace, rehabilitate, or acquire the equivalent of the injured resources [9–13], typically accomplished through primary and compensatory restoration.

Within an injured area, primary restoration is intended to restore resources and services to, or as close as is practical to, the baseline condition, defined as what would exist but for the injury [14,15]. In the context of injured coral reefs, primary restoration approaches have included righting and reattaching dislodged colonies, stabilizing loose rubble to prevent additional injury, transplanting colonies to the injured site from adjacent donor areas or nurseries, and reengineering of the injured reef framework [16]. These techniques and others have been applied to U.S. trust resources in Florida [4,17–20], U.S. Virgin Islands [4,5], Puerto Rico [2], American Samoa [21,22], Hawaii [7,22], and other U.S.-affiliated Pacific Islands [7,22].

Prompt and effective primary restoration is intended to reduce the loss of natural resources and their services between the time of an incident and the time at which the injured resources return to baseline. In practice, however, this time period is often lengthy, resulting in an interim net service loss (where services are defined in an ecological context as outcomes resulting from biophysical processes within an ecosystem [23] and in an NRDA context as natural resource functions that benefit another natural resource and/or the public [24]). In addition, if the injured resource cannot be restored fully to baseline conditions, there is a service loss in perpetuity. Compensatory restoration is intended to address these interim or perpetual service losses. In the context of coral reef injuries, compensatory restoration usually occurs at a location other than the injured site, and common projects include the creation of artificial reefs [16,25] and restoration of injured areas for which there is no identified or financially viable responsible party. Compensatory restoration projects have also included out-of-kind actions, such as marine debris removal, invasive species control, coral nurseries, improvement of the injured site above baseline conditions, or installation of navigation beacons [1] to prevent future resource injury.

Quantifying and valuing the amount of service loss can be challenging, particularly for coral reef injuries. Comprehensive measurement of ecosystem services is not straightforward in practice, and determination of equivalent services from either scientific or social standpoints is even less so [26]. Some ecosystem services are difficult to measure at all, while others can be measured in various meaningful ways [26,27]. Measuring ecosystem services is further complicated by the fact that they can be provided at multiple spatial scales in the environment, requiring widely varying assessment techniques and statistical considerations. Coral reefs provide extensive ecosystem services [17,28–31]. Valuation techniques for coral reef ecosystem services vary widely [28,32–38], and different valuation methods can produce very different results [37]. Unlike ecosystem goods – which are harvests of environmental products for direct human use [39] – such as subsistence or recreational fisheries – many ecosystem services never pass though an economic open market to obtain an actual market value. Due to these complications, most injury assessment studies use indicators of ecological services [40] rather than measuring ecosystem services directly. Thus, in an NRDA, the quantity of compensatory restoration required to offset service losses as a result of an injury is not calculated by directly measuring a comprehensive suite of ecosystem services, but is instead often determined by habitat equivalency analysis (HEA) or resource equivalency analysis (REA) [26,41–44]. REA is a resource-to-resource method that references the number of organisms lost and gained, and HEA is a service-to-service method that references habitat area lost and gained [42]. The two methods are algebraically identical and are used to calculate the quantity of compensatory restoration that will generate natural resource services equivalent to service losses due to an injury [24,26,41–44]. Service losses and compensatory benefits are quantified in non-monetized units such as discounted service-acre-years (DSAYs). One DSAY represents the suite of baseline services provided by an acre of the injured habitat at baseline conditions in the base year [15].

Using HEA, the amount of interim and perpetual service losses for an injury is measured as the area under a recovery curve, bounded by the baseline and the projected recovery [15]. This trajectory showing projected recovery over time is required for the HEA calculations [15,17,45]. Recovery estimates may be based on best professional judgment, literature values, or recovery modeling. In the absence of empirical data to either plot or simulate recovery, many cases rely on expert opinion to determine recovery time. Changes in the estimated recovery rate will change the shape of the curve, thus changing the area under the curve and the amount of compensatory restoration required (Fig. 1). However, with the HEA discount rate of 3% commonly used in NRDAs, losses or benefits that extend beyond 70–100 years have very little bearing on the restoration requirement because they are so heavily discounted [46].

A key assumption of HEA is that the services lost due to the injury and those provided by the compensatory restoration are comparable in terms of type, quality, and value [15]. Therefore, HEA application requires that practitioners be able to measure in a common metric the ecosystem services provided by a natural resource or habitat and a compensatory project. This metric is intended to be representative of the entire (yet often undefined) suite of ecosystem services being provided. It is usually assumed that recovery of the service directly captured by the metric will be

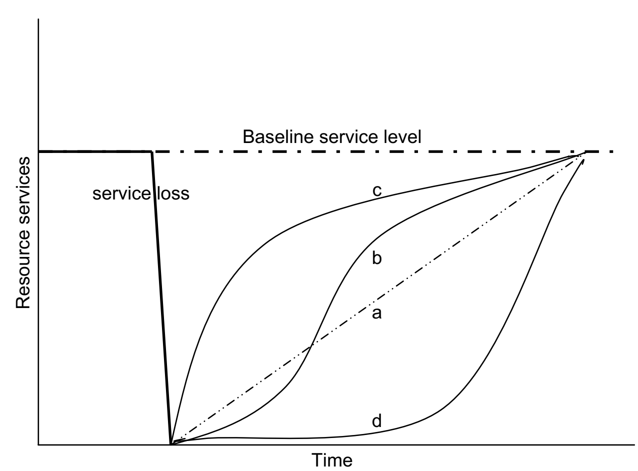

Fig. 1. HEA diagram showing service loss resulting from an injury decreasing the amount of baseline services provided by a habitat. Estimates assume a linear resource recovery (a). Trajectories (b)–(d) illustrate variable recovery rates that are more ecologically variable than a linear function. More immediate recovery (c) has a higher influence due to HEA discounting than either recovery that passes through a transition state (b) or longer term recovery (d).

accompanied by restoration of all other ecosystem services. In essence, a proxy is selected to represent the full suite of ecosystem services, and serves as the measure of restoration success. For example, in the Florida Keys National Marine Sanctuary, seagrass HEAs use aboveground seagrass biomass as the common metric since this measure is highly correlated with services provided by the habitat [45]. However, the assumption of a representative metric is challenging in complex ecosystems [47], and the successful application of a metric in seagrass ecosystems required extensive supporting research to allow selection of a simple representative. Coral reefs have not yet undergone a similar process where metrics are parsed for uniform application under HEA and are more complex in terms of ecosystem services provided. Services can also occur across habitat types, so the ecological context of a coral reef within surrounding mangrove, seagrass, soft and hard bottom environments, and open water habitats could also be considered (with accordingly increased complexity). Nonetheless, when a single metric is not applicable across habitat or project types, conversion factors may be used to translate measurement of one component's services into equivalents of the other component.

The applicability of a specific metric depends upon landscape context, biophysical context, and the capacity of services being provided, including ecological scale, type and extent of injury and defined goals for restoration success [15,42,48]. As with any ecosystem, there may be no single universal metric applicable to all coral reef NRDA situations, in part because a metric must fill a variety of roles, yet it may be feasible to capture a sufficient suite of services in one or two practical metrics. Ideally, the NRDA metric would link the type of information collected during the field damage assessment, the approach for defining, quantifying, and projecting recovery from the injury, the available options for compensatory restoration projects appropriate to the injury, and recovery monitoring protocols. For HEA purposes, a metric should represent ecosystem services to address specific recovery goals, provide appropriate and meaningful measurements to determine a realistic recovery horizon that is representative of the injured habitat, show measurable change over time at the organizational level (i.e. organismal, landscape) at which the injury occurred, and be quantifiable using repeatable, field-realistic, cost-effective damage assessment techniques that allow reliable return assessments as well. In addition, a metric should be applicable to a compensatory restoration project as the same type and quality service, or translatable to another service through peer-reviewed and broadly accepted conversion factors. Finally, the metric should be appropriate for monitoring restoration progress with the sensitivity to detect whether mid-course corrections are needed.

2. Coral reef NRDA metrics

Most HEA applications in coral reef environments have historically used one of several metrics. In this section, we examine these metrics for the robustness, flexibility, and application within HEA to predict complex and compound (larger scale) results from the original injury. Improving coral reef NRDA practice will likely require trustees to move beyond the relatively simple metrics that have historically been used toward a more holistic system of metrics that may more accurately reflect the complexity of coral reef ecosystems. We do not imply that historical applications have been in error, but rather suggest possible directions for future discussion and refinements.

2.1. Single, total coral cover metric

For coral reef grounding injuries in the U.S., NRDAs have traditionally used a two-dimensional measurement of all biological coral tissue cover measured as either area or percent cover [17]. The conceptual basis is that an increase in total coral cover requires successful recruitment and growth, and will promote reef structural complexity and ecosystem richness [17]. The advantage of a coral cover metric in the NRDA process is that the service flow is intuitive; the amount of total coral cover injured requires that a similar amount of coral cover be restored. Field measurements of benthic cover for the initial injury assessment and recovery monitoring can be relatively straightforward [49,50] (although less so for branching corals). In addition, coral cover has been a common parameter in reef monitoring publications for the past several decades, so landscape-scale historical data may exist for a particular reef. From an economic perspective, a coral cover metric is a transparent application of the HEA equation: the amount of coral cover inside the injury is expressed as a percentage of coral cover in a reference area (selected to represent the baseline condition of the injured area), and this proportion is projected over a time to estimate coral cover recovery. With a single coral cover metric that treats all coral species equally, no weighting factors are required within the HEA to equate for different levels of service contributions by different coral species. A coral cover metric is easily translatable to compensatory restoration projects such as transplantation, coral nurseries, and recruitment seeding that are designed to increase coral cover. Therefore, ecological service conversion factors for compensatory restoration options are not required to account for a compensatory restoration that differs from the injury metric.

From an ecological perspective, however, a coral cover metric may be an overly simplistic representative of ecosystem services. A coral cover metric requires the assumption that scleractinian coral cover is correlated with other services provided by a coral reef (such as those provided by non-scleractinian sessile benthic invertebrates, mobile invertebrates, vertebrate herbivores and carnivores, algae, and the nonliving reef framework itself). While this assumption may be supportable for certain types of injuries or reefs, for others, particularly those with low pre-injury scleractinian coral cover, ecosystem services may be more influenced by other benthic organisms such as sponges, octocorals, or algae. In these habitats, other taxa may recover more or less quickly than scleractinian corals, and the restoration requirement may be biased if hard coral cover is the only metric used.

A coral cover metric also does not address variations in ecosystem services provided by different coral species or functional groups (e.g. diversity, composition, colony size, morphology, potential accretion rate, level and type of habitat provided) nor whether services scale with size or age (e.g. reproduction). An HEA recovery estimate for a coral cover metric may be based on recovery of total coral cover or on attributes of selected species, such as for a species-oriented recovery model [51]. With a species-oriented approach, the selected species needs to have estimated growth rates, morphology and other species attributes appropriate to the context of the injury and the reef, for these will affect recovery projections and restoration planning. A coral cover metric therefore has the potential to over- or under-represent the contributions of selected species attributes. This could become complicated within an NRDA framework if a species listed as threatened or endangered under the U.S. Endangered Species Act (such as Caribbean acroporids) is involved, but is not a dominant species at the injury site.

The limitations inherent with a metric of two-dimensional area of total living scleractinian coral tissue imply that this metric should be limited to specific types of injuries or types of coral reef communities rather than broadly applied to all injuries. A coral cover metric could be appropriate for an injury that does not directly address structural complexity, such as an abrasion or other tissue injury to a coral colony, or an injury to a hard bottom or low-relief habitat. A two-dimensional, total coral cover metric would be best used on reefs dominated by scleractinian corals of similar species or functional groups providing similar ecosystem services. Thus, a coral cover metric could be applicable to early successional communities [52] that are dominated by short-lived coral species with high recruitment rates and small colony size (e.g. *Porites astreoides* in the Florida Keys [53,54]).

2.2. Composite metrics using percent cover

Composite metrics could be used to capture changes in services due to an injury and would retain the NRDA's focus on the habitat-level assessment. Composite metrics have been suggested as an alternative to a two-dimensional, total coral cover metric in order to more comprehensively account for coral reef community diversity [55]. Losses associated with coral cover of one or multiple coral species can be aggregated with cover measurements of sponges, algae, or other habitat providing organisms. The additional community information required for composite metrics may involve more field data collection (and associated assessment costs) for both the injury assessment and for recovery monitoring than a single, total coral cover metric, although in many cases these data are already collected as part of current protocols.

When composite metrics are used within an HEA to assess habitat injury, either individual metrics could be aggregated and weighted prior to the HEA, or separate HEA equations could be calculated for each individual metric and weighted afterwards. For either approach, options for weighting include relative cover [55] or expert opinion of the relative contribution of each to the local ecosystem's total services. Composite metrics used in environments other than coral reefs have historically been weighted and aggregated into a single HEA equation to represent the total percentage of service loss caused by the injury [42]. The recovery projection is based on the recovery of the metrics as a whole, rather than the recovery of each component which, in coral reef environments, may give greater recovery importance to fast-growing non-coral species, such as algae or sponges, than to scleractinian corals. To decrease this possibility, additional consideration needs to be given to weighting the metrics within the HEA equation. Another approach would be to use several individual metrics in individual HEA equations [55]. This enables simultaneous comparison of services among different biological components. As an example, one may calculate the two-dimensional area of scleractinian corals, octocorals, and sponges that were lost, project the recovery of each separately, and proportionally weight metrics accordingly. Calculating an HEA for each metric individually and then weighting results afterwards would allow for variation in recovery rate among component categories, thereby reducing the possibility that fast-growing taxa would disproportionally contribute to recovery.

Composite metrics approaches have several challenges in a coral reef environment. While they recognize that scleractinian corals are not universally appropriate as indicator species, they do not address intraspecific variation in services that may be non-linearly correlated with size, such as the capacity of branching corals to act as habitat refuges. In addition, the increased specificity of composite metrics may reduce the suite of available compensatory options. If specific compensatory options are not available to address each factor within the composite metric, conversion factors may be required to account for the variation in services provided by the restoration versus the injury. In order to quantify the potential to regain services with each compensatory restoration project under consideration, the recovery of each metric component is projected over time for each compensatory project as well as for the injury.

2.3. Size-frequency distributions

Size-frequency distributions at the species or functional group level can reflect the life history strategies of different corals [56,57], have predictive power for population development [58], and allow representation of the (typically non-linear) relationship between services and colony size, thus providing insights into ecological function. Determining size-frequency distributions of habitat-forming organisms (in this case, corals) is conceptually straightforward. Each colony present is measured and assigned to a size class and species or functional group. The number of classes and the size range within each class should be appropriate for the services provided by different life history stages for the species of interest. For example, a 5 cm coral might be a juvenile for one species but a fully mature, reproductive adult for another species. Coral size-frequency distributions have been used to examine the effects of bleaching [59], disease [60], lesions [61,62], marine protected area creation [63], hurricanes [64,65], and water quality degradation [56,66]. The size-frequency method is also beginning to be applied to coral recovery monitoring from vessel groundings [5,67,68], and species-specific recovery modeling [67].

A size-frequency distribution metric is a shift from a habitat-based HEA assessment to a more detailed resource-based REA framework. This allows for quantification of service flows for a greater number of distinct service types. Injury quantification for multiple coral taxa addresses the aforementioned variability in functions, such as refugia quality among scleractinian coral species or functional groups, and acknowledges that different coral species provide different services based on life histories, morphology, and relative abundance within the affected community. In addition, size-frequency distributions may also be used to capture specific ecological services provided by coral reef organisms other than scleractinian corals (e.g. sponges, octocorals). A more detailed injury quantification provides additional community information yet would not require habitat-level information weighting (see above). An REA framework has been used in non-coral reef environments for capturing services that are provided by ecosystem components such as mobile vertebrates or invertebrates, which, while not habitat-forming, also provide ecosystem services [69].

A size-frequency distribution metric has several limitations, however. First, the additional effort to obtain the requisite detailed data can require significantly more time (and cost) for data collection and analysis than coral cover-based metrics. Second, size classes are typically defined by the longest axis of a colony. This potentially poses a problem of comparative geometry among, for example, non-hemispherical species and species with multiple growth forms – a 20 cm diameter coral head and an encrusting colony 20 cm wide by 2 cm long might be assigned to the same size class despite vastly different surface areas (and concomitant resource services). In such cases, it may be preferable to use the average length of colony growth axes, or to use area-frequency rather than size-frequency. Third, size-frequency may not be easily applied to injuries that have significant partial mortality. Moving an injured colony to a smaller size class may not necessarily reflect the non-linear relationship between size class and service loss. For example, tissue regrowth from an abrasion injury may occur at a different rate than regrowth from an injury that includes skeletal loss. Also, partial mortality from fragmentation would likely have non-linear effects on ecosystem services. Finally, because coral abundance typically recovers faster than coral cover after a grounding [70], recovery horizons based on size-frequency may differ drastically from those based on coral cover.

While measuring losses and gains of organisms rather than habitat is a comprehensive method for damage assessment and recovery monitoring, a resource-based REA method such as size-frequency distribution also has inherent challenges for scaling to compensatory restoration. Rather than estimating the primary recovery and compensatory restoration benefits for a single species, this method requires calculating those parameters for multiple size classes within multiple scleractinian species (or genera or functional groups) and for organisms other than scleractinian corals. Size-frequency information can be more difficult than coral cover to use in an HEA or REA which requires an estimate of recovery time and the shape of the recovery curve. Comparison of size-frequency distributions via goodness-of-fit tests at each time step allows an estimate of how long it takes for the populations to converge [67] but the statistics can only discern whether or not the populations are the same, and provide little information on the shape of the recovery curve, a critical parameter in HEA and REA. Alternatively, recovery projections could be modeled for each size category within each coral species/genera/group and then combined across each species/genera/group. Despite its advantage in capturing a greater suite of services by multiple community components, this approach may require ecological service conversion factors between size classes of a single species and between different species to fit with available compensatory restoration options. Lack of objective and quantitative methods to develop these REA conversion factors is a substantial obstacle to broad implementation of this approach. However, current coral cover and composite metrics HEA approaches also face identical challenges (see above).

For NRDAs in non-coral reef environments, scaling to compensatory restoration has been addressed using a habitat-based replacement cost (HRC) approach which combines the HEA and REA concepts [71,72]. In this method, losses are first quantified using an REA. Next, suitable restoration options are identified to address habitat limitations for each impacted species group, prioritized by quantification of benefits, and selected for each impacted species. Finally, restoration options are scaled so that increases in production as a result of restoration options would offset losses for each impacted species group [71]. In oil spill NRDAs, common examples of this approach include compensating for bird, fish, or shellfish losses with creation of marshes to provide nesting, nursery and foraging habitat [73,74]. The HRC approach may be appropriate for coral reef REAs; however, it does assume habitat is the limiting factor for local production of replacement organisms. For certain coral reefs and coral reef injuries, coral recruitment may be the limiting factor rather than available and appropriate habitat, which would pose problems in developing comparative recovery functions.

An intensive REA approach is likely best suited for comparatively large incidents in complex settings. For large incidents, the potential increase in precision afforded by the REA approach may lead to an increased level of confidence that the calculated restoration requirement is sufficient to fully offset the losses due to injury. For smaller incidents or those that occur in areas that have relatively little species diversity, the common NRDA method of using a single representative metric and HEA may be sufficient.

2.4. Topographic complexity

One of the criticisms of two-dimensional metrics such as coral cover or size-frequency distribution is that none adequately capture a reduction in the complex three-dimensional reef topology. A reduction, restructuring, or destabilization of topographic complexity (e.g. pulverized reef spur) can limit coral recruitment and recovery [5,75], can shift the recovering community composition [76,77], can increase vulnerability to storm damage [75], and can influence reef hydrodynamics which could lead to an expansion of the original injury. Although injuries to the physical reef structure are frequently measured in damage assessments, it has not traditionally been included within HEA equation itself. Instead, reef framework injury has been included within the NRDA process as an ancillary service, a service integral to the natural resource at the landscape scale [10]. As such, it influences the criteria for determining compensatory restoration options appropriate to the injury. Within a suite of compensatory restoration options identified for a reef injury involving structural complexity, those that restore any physical structure have been considered more favorable to implement than those not restoring structure. This approach does not include relating the quantity of complexity lost to what will be replaced.

Services associated with a complex, three-dimensional structure of the reef could be captured with a metric and included in the HEA equation. While there are many instances, particularly in Florida and the Caribbean, when topographic complexity would be unlikely to recover naturally in the limited HEA-relevant time frame of 70–100 years and should be considered an injury in perpetuity for HEA purposes, this assumption will not be appropriate for relatively fast-growing coral species. Monitoring framework structure is intended to evaluate whether the structure is present and intact, and, if coupled with a biological metric (coral cover or size-frequency), can evaluate the ecological contributions of the framework to the system. A topographic complexity metric has a straightforward translation to compensatory restoration projects. Specific approaches to accelerate habitat restoration using reef engineering techniques have varied widely and have included spur rebuilding, modules, and reef crowns [16]. The design and composition of the engineered structure, however, will affect its potential for successful coral recruitment [54,77] and growth, important factors to be considered to ensure coral growth on any engineered structure so granting credit for restoration of framework structure may become a secondary benefit in the case of an injury in an area of limited or declining recruitment. Therefore, incorporating a topographic complexity metric with a cover metric remains an unresolved challenge for further research in coral reef restoration.

3. Coral reef NRDAs: unstable baselines and multiple stable states

3.1. Baselines

In NRDA recovery projections, the baseline projection is generally assumed to be static (Fig. 2) [26]. In essence, this means the

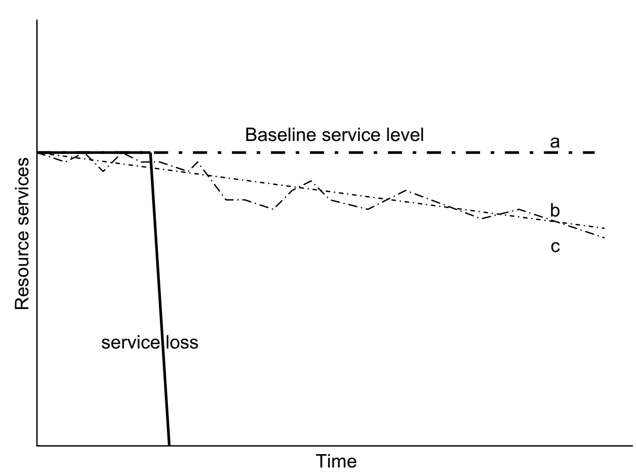

Fig. 2. Changing baseline service levels of the injured reef and uninjured reference populations may need to be considered when recovery trajectories are projected. A steady decline may be modeled (b) or, data may show a variable decline over time (c).

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baseline, the condition that would exist but for the injury, is considered equivalent to the pre-injury condition throughout the HEA time frame. Within the HEA recovery projections, this does not allow for a shifting baseline caused by changes in the condition of the reference community (Fig. 2). In an NRDA context, if the baseline is not strictly re-established, different services will be provided instead. In most cases, a static baseline for recovery projections is necessitated by a lack of adequate data. However, one approach to determine when the recovery projection meets the baseline projection could be to model a reference community simultaneously with the injured community [67] and re-adjust the recovery scenario periodically (although this has severe practical limitations in a legal case). This type of simultaneous recovery modeling further requires the assumption that the population dynamics are equivalent in the injured and uninjured areas.

3.2. Multiple stable states

After a disturbance such as a vessel grounding, a coral reef may recover temporarily or permanently to an alternate state rather than via classical succession to the pre-injury condition [78–81]. In such a situation, the coral species or community structure lost due to the injury would not be equivalent to what actually comes back (Fig. 3). Unrestored framework injury can contribute to the development of a community far different from that of the pre-injury community, such as in the case of the M/V Wellwood in the Florida Keys, in which recovery of a pulverized spur and groove reef stabilized in a low-relief hard bottom state until engineered structure was added [82]. Similarly, a coral-dominated reef could become an algal dominated reef [79], or recovering communities may be composed of different coral species which may provide different services. For example, the limited natural coral recruitment into injuries in the Florida Keys and the Caribbean is dominated by brooding coral species, such as *P. astreoides*, with weedy growth characteristics (rapid growth, limited accretion rate, relatively short lifespan) rather than framework-building corals that have slower growth but more potential to provide structural habitat over a longer time [53,77,83–85]. Complete recovery to the baseline condition (if possible) may simply be outside of the HEA-relevant time frame. Alternate stable states may be addressed in the HEA either as a loss in perpetuity or through conversion factors to translate the ecological services provided by the new community into quantifiable units of the original communities. These conversion factors would ideally be based on objective measurements of the services provided by both the original and re-established coral communities. In practice, however, such conversion factors are often predicated on expert opinion due to lack of available data.

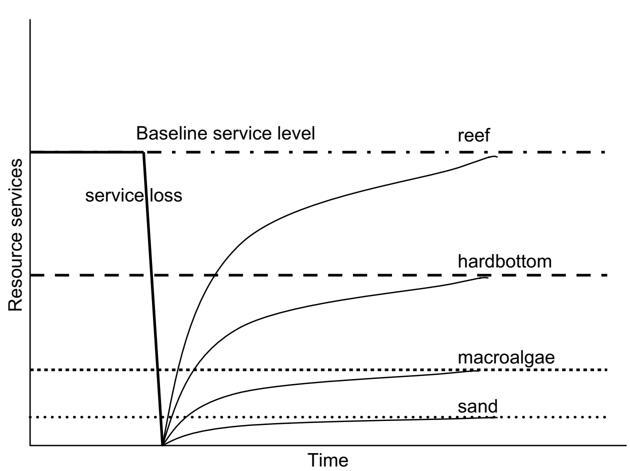

Fig. 3. An injured reef may not recover to its initial state, but may instead recover to an alternative state that will not provide the type or level of ecosystem services that were lost.

4. Conclusions

Coral reef restoration is very much a developing science. Few, if any, injuries have been followed from impact to complete recovery as part of the NRDA process. Consequently, expert estimates about whether a site will recover in 30, 50, 300 years, or not at all, are necessarily imperfect [17], but bear the responsibility of being the best available information at present. Almost all of the approaches detailed in our review rely heavily on expert opinion, which is unlikely to be determined in a universally accepted manner, contributing to the adversarial nature of determining the extent and cost of restoration. Thus, this review is also an encouragement for coral reef NRDAs to become a process that supports the development of objective (quantitative) rather than the current, often subjective process. As more informative data emerge from research, restoration monitoring, and HEA applications, they should be applied to advance the NRDA process in conjunction with coral reef restoration science.

In its simplest form, the objective of coral reef restoration conducted through the NRDA process is to restore the services lost from the injuries caused by the responsible party. It is often difficult to know whether the trustee actions are sufficient to reach this objective given the current state of reef restoration science and NRDA practice. While the practical and measurable goals of restoration are to rapidly re-create the structure and functions of an injury habitat, the approaches for realizing this goal are continually evolving. There is a delicate balance between broad, general operating principles and site specificity. Careful selection of the theoretical NRDA approach (HEA-based using two-dimensional coral cover or composite metrics, or REA-based using size-frequency distributions) and metrics appropriate to both the degree and extent of injury and of habitat type will serve as a vital link between the damage assessment, recovery modeling, compensatory calculations, and recovery monitoring. An immense amount of information is necessary to fully understand the type and magnitude of ecological services provided by the injured coral reef in its baseline condition, the manner in which those ecological services will recover following the injury, and the relationship of those services with those provided via compensatory restoration projects. Our challenge is to capture this information in broadly accepted metrics that are cost-effective to obtain. However, more complete understanding of coral reef ecological services is required to objectively determine whether selected compensatory restoration projects adequately restore lost services for a given injury.

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References

[1] Schmahl GP, Deis DR, Shutler SK. Cooperative natural resource damage assessment and coral reef restoration at the container ship *Houston* grounding

in the Florida Keys National Marine Sanctuary. In: Precht WF, editor. Coral reef restoration handbook. Boca Raton, Florida: CRC Press; 2006. p. 235–56.

[2] Bruckner AW, Bruckner RJ. Restoration outcomes of the *Fortuna Reefer* grounding at Mona island, Puerto Rico. In: Precht WF, editor. Coral reef restoration handbook. Boca Raton, Florida: CRC Press; 2006. p. 257–69.

[3] Hudson JH, Diaz R. Damage survey and restoration of M/V Wellwood grounding site, Molasses Reef, Key Largo National Marine Sanctuary, Florida. Proceedings of the Sixth International Coral Reef Symposium, Australia, vol. 2; 1988. p. 231–6.

[4] Jaap WC. Coral reef restoration. Ecological Engineering 2000;15(3–4):345–64.

[5] Rogers CS, Garrison VH. Ten years after the crime: lasting effects of damage from a cruise ship anchor on a coral reef in St. John, U.S. Virgin Islands. Bulletin of Marine Science 2001;69(2):793–803.

[6] National Oceanic and Atmospheric Administration, Office of Response and Restoration. Oil spills in coral reefs. Planning and response considerations. Silver Spring, Maryland: National Oceanic and Atmospheric Administration; 2001.

[7] Jokiel PL, Kolinski SP, Naughton J, Maragos JE. Review of coral reef restoration and mitigation in Hawaii and the U.S.-affiliated Pacific islands. In: Precht WF, editor. Coral reef restoration handbook. Boca Raton, Florida: CRC Press; 2006. p. 271–90.

[8] Tetra Tech EC Inc. Maritime Industry and Coastal Construction Impacts Workshop. Dania Beach, Florida: Southeast Florida Coral Reef Initiative, Maritime Industry and Coastal Construction Impacts Focus Team; 2007.

[9] National Marine Sanctuaries Act. 1972, 16 U.S.C. § 1443.

[10] Oil Pollution Act. 33 U.S.C. 1990, § 2701, et seq.

[11] Comprehensive Environmental Response, Compensation, and Liability Act. 1980, 42 U.S.C. § 9607.

[12] National Park Service Resource Protection Act. 1990, 16 U.S.C. § 19jj.

[13] Jones CA, Pease KA. Restoration-based compensation measures in natural resource liability statutes. Contemporary Economic Policy 1997;15(4): 111–22.

[14] Thayer GW, McTigue TA, Bellmer RJ, Burrows FM, Merkey DH, Nickens AD, et al. Science-based restoration monitoring of coastal habitats. In: A framework for monitoring plans under the Estuaries and Clean Waters Act of 2000 (Public Law 160-457), vol. 1. NOAA Coastal Ocean Program Decision Analysis Series No. 23. Silver Spring, Maryland: NOAA National Centers for Coastal Ocean Science; 2003.

[15] National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program. Habitat equivalency analysis: an overview. Revised 2006. Silver Spring, Maryland: National Oceanic and Atmospheric Administration; 1995.

[16] Zimmer B. Coral reef restoration: an overview. In: Precht WF, editor. Coral reef restoration handbook. Boca Raton, Florida: CRC Press; 2006. p. 39–59.

[17] Shutler SK, Gittings S, Penn T, Schittone J. Compensatory restoration: how much is enough? Legal, economic, and ecological considerations. In: Precht WF, editor. Coral reef restoration handbook. Boca Raton, Florida: CRC Press; 2006. p. 77–93.

[18] Bourque A, Clark R, Curry R, Lanzendorf B, Carriero J, Whittington T, et al. Allie B grounding site restoration plan/environmental assessment. Homestead, Florida: National Park Service, U.S. Department of the Interior, Biscayne National Park; 2007.

[19] Bourque A, Whittington T, Lanzendorf B, Carriero J, Tilmant J. Igloo Moon grounding site restoration plan/environmental assessment. Homestead, Florida: U.S. Department of the Interior, National Park Service, Biscayne National Park; 2007.

[20] Carson D, Vare C. Evaluation of reef restoration efforts off Boca Raton, Florida. West Palm Beach, Florida: Palm Beach County Environmental Resources Management; 1993.

[21] Government of American Samoa, U.S. Department of the Interior, National Oceanic and Atmospheric Administration. Emergency Restoration Plan and Environmental Assessment Pago Pago Harbor American Samoa. Government of American Samoa, U.S. Department of the Interior, National Oceanic and Atmospheric Administration; 1999.

[22] Bentivoglio A. Compensatory mitigation for coral reef impacts in the Pacific Islands. Final Report. Honolulu, Hawaii: United States Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office; 1993.

[23] Egoh B, Rouget M, Reyers B, Knight AT, Cowling RM, van Jaarsveld AS, et al. Integrating ecosystem services into conservation assessments: a review. Ecological Economics 2007;63:714–21.

[24] Thur SM. Refining the use of habitat equivalency analysis. Environmental Management 2007;40:161–70.

[25] Spieler RE, Gilliam DS, Sherman RL. Artificial substrate and coral reef restoration: what do we need to know to know what we need? Bulletin of Marine Science 2001;69(2):1013–30.

[26] Mazzotta MJ, Opaluch JJ, Grigalunas TA. Natural resource damage assessment: the role of resource restoration. Natural Resource Journal 1994;34:153–78.

[27] Hatcher BG. Coral reef ecosystems: how much greater is the whole than the sum of the parts? Coral Reefs 1997;16:S77–91.

[28] Costanza R, d'Arge R, de Groot RS, Farber S, Grasso M, Hannon B, et al. The value of the world's ecosystem services and natural capital. Nature 1997;387:253–60.

[29] Moberg F, Folke C. Ecological goods and services of coral reef ecosystems. Ecological Economics 1999;29:215–33.

[30] Souter DW, Linden O. The health and future of coral reef systems. Ocean and Coastal Management 2000;43:657–88.

[31] Nyström M, Folke C. Spatial resilience of coral reefs. Ecosystems 2001;4: 406–17.

[32] Spurgeon JPG. The economic valuation of coral reefs. Marine Pollution Bulletin 1992;24(11):529–36.

[33] Cesar HSJ, van Beukering PJH. Economic valuation of the coral reefs of Hawai'i. Pacific Science 2004;58(2):231–42.

[34] Wielgus J, Chadwick-Furman NE, Zeitouni N, Shechter M. Effects of coral reef attribute damage on recreational welfare. Marine Resource Economics 2003;18:225–37.

[35] van Beukering PJH, Cesar HSJ. Ecological economic modeling of coral reefs: evaluating tourist overuse at Hanauma Bay and algae blooms at the Kihei Coast, Hawai'i. Pacific Science 2004;58(2):243–60.

[36] Anderson JEC. The recreational cost of coral bleaching – a stated and revealed preference study of international tourists. Ecological Economics 2007;62: 704–15.

[37] Brander LM, Van Beukering P, Cesar HSJ. The recreational value of coral reefs: a meta-analysis. Ecological Economics 2007;63(1):209–18.

[38] Ahmed M, Umali GM, Chong CK, Rull MF, Garcia MC. Valuing recreational and conservation benefits of coral reefs – the case of Bolinao, Philippines. Ocean and Coastal Management 2007;50:103–18.

[39] Daily GC, Alexander S, Ehrlich PR, Goulder L, Lubchenco J, Matson PA, et al. Ecosystem services: benefits supplied to human societies by natural ecosystems. Issues in Ecology 1997;2:1–16.

[40] Ruiz-Jaen MC, Aide TM. Restoration success: how is it being measured? Restoration Ecology 2005;13(3):569–77.

[41] Unsworth RE, Bishop RC. Assessing natural resource damages using environmental annuities. Ecological Economics 1994;11:35–41.

[42] Allen II PD, Chapman DJ, Lane D. Scaling environmental restoration to offset injury using habitat equivalency analysis. In: Bruins RJF, Heberling MT, editors. Economics and ecological risk assessment: applications to watershed management. Boca Raton, Florida: CRC Press; 2005. p. 165–84.

[43] Hampton S, Zafonte M. Calculating compensatory restoration in natural resource damage assessments: recent experience in California. Proceedings of the California and the World Ocean '02 Conference: revisiting and revising California's ocean agenda. Reston, Virginia; 2005. p. 833–44.

[44] Zafonte M, Hampton S. Exploring welfare implications of resource equivalency analysis in natural resource damage assessments. Ecological Economics 2007;61:134–45.

[45] Fonseca MS, Julius BE, Kenworthy WJ. Integrating biology and economics in seagrass restoration: How much is enough and why? Ecological Engineering 2000;15:227–37.

[46] National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program. Discounting and the treatment of uncertainty in natural resource damage assessment. Silver Spring, Maryland: National Oceanic and Atmospheric Administration; 1999.

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

[48] King DM. Comparing ecosystem services and values: with illustrations for performing habitat equivalency analysis. Damage Assessment and Restoration Program. Silver Spring, Maryland: National Oceanic and Atmospheric Administration; 1997.

[49] Banks K, Dodge RE, Fisher L, Stout D, Jaap W. Florida coral reef damage from nuclear submarine grounding and proposed restoration. Special issue. Journal of Coastal Research 1998;26:64–71.

[50] Hudson JH, Goodwin WB. Assessment of vessel grounding injury to coral reef and seagrass habitats in the Florida Keys National Marine Sanctuary, Florida: protocol and methods. Bulletin of Marine Science 2001;69(2):509–16.

[51] Piniak GA, Fonseca MS, Kenworthy WJ, Whitfield PE, Fisher G, Julius BE. Applied modeling of coral reef ecosystem function and recovery. In: Precht WF, editor. Coral reef restoration handbook. Boca Raton, Florida: CRC Press; 2006. p. 95–117.

[52] Zengel S, Hinkeldey H. Coral reef recovery: literature review and recommendations for damage assessment and restoration planning. Silver Spring, Maryland: Research Planning, Inc.; 2001.

[53] Miller MW, Weil E, Szmant AM. Coral recruitment and juvenile mortality as structuring factors for reef benthic communities in Biscayne National Park, USA. Coral Reefs 2000;19:115–23.

[54] Moulding AL. Coral recruitment in the Florida Keys: patterns, processes, and applications to reef restoration. Coral Gables, FL: University of Miami; 2007. 157pp.

[55] Milon JW, Dodge RE. Applying habitat equivalency analysis for coral reef damage assessment and restoration. Bulletin of Marine Science 2001;69(2):975–88.

[56] Meesters EH, Hilterman M, Kardinaal E, Keetman M, deVries M, Bak RPM. Colony size-frequency distributions of scleractinian coral populations: spatial and interspecific variation. Marine Ecology Progress Series 2001;209:43–54.

[57] Vermeij MJA, Bak RPM. Species-specific population structure of closely related coral morphospecies along a depth gradient (5–60 m) over a Caribbean reef slope. Bulletin of Marine Science 2003;73(3):725–44.

[58] Bak RPM, Meesters EH. Coral population structure: the hidden informaiton of colony size-frequency distributions. Marine Ecology Progress Series 1998;162:301–6.

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[59] Mumby PJ, Chisholm JRM, Edwards AJ, Clark CD, Roark EB, Andrefouet S, et al. Unprecedented bleaching-induced mortality in *Porites* spp. at Rangiroa Atoll, French Polynesia. Marine Biology 2001;139:183–9.
[60] Richardson LL, Voss JD. Changes in a coral population on reefs of the northern Florida Keys following a coral disease epizootic. Marine Ecology Progress Series 2005;297:147–56.
[61] Meesters EH, Wesseling I, Bak RPM. Partial mortality in three species of reef-building corals and the relation with colony morphology. Bulletin of Marine Science 1996;58(3):838–52.
[62] Ruesink JL. Coral injury and recovery: matrix models link process to pattern. Journal of Experimental Marine Biology and Ecology 1997;210:187–208.
[63] Epstein N, Vermeij MJA, Bak RPM, Rinkevich B. Alleviating impacts of anthropogenic activities by traditional conservation measures: can a small reef reserve be sustainedly managed? Biological Conservation 2005;121: 243–55.
[64] Beltràn-Torres AU, Muñoz-Sànchez L, Carricart-Ganivet JP. Effects of Hurricane Keith at a patch reef on Banco Chinchorro, Mexican Caribbean. Bulletin of Marine Science 2003;73(1):187–96.
[65] Mumby PJ. Bleaching and hurricane disturbances to populations of coral recruits in Belize. Marine Ecology Progress Series 1999;190:27–35.
[66] Lewis JB. Abundance, distribution and partial mortality of the massive coral *Siderastrea siderea* on degrading coral reefs at Barbados, West Indies. Marine Pollution Bulletin 1997;34(8):622–7.
[67] Lirman D, Miller MW. Modeling and monitoring tools to assess recovery status and convergence rates between restored and undisturbed coral reef habitats. Restoration Ecology 2003;11(4):448–56.
[68] Hudson JH, Schittone J, Anderson J, Franklin EC, Stratton A. M/V Alec Owen Maitland coral reef restoration monitoring report, monitoring events 2004–2007. Florida Keys National Marine Sanctuary Monroe County, Florida. Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Sanctuary Program; 2008.
[69] Sperduto MB, Powers SP, Donlan M. Scaling restoration to achieve quantitative enhancement of loon, seaduck, and other seabird populations. Marine Ecology Progress Series 2003;264:221–32.
[70] Gittings SR, Bright TJ, Hagman DK. The M/V Wellwood and other large vessel groundings: coral reef damage and recovery. In: Ginsburg RN, editor. Global aspects of coral reefs. Health, hazards, and history. Miami, Florida: University of Miami; 1993. p. 174–80.
[71] Allen II PD, Raucher R, Strange E, Mills D, Beltman D. The habitat-based replacement cost method: building on habitat equivalency analysis to inform regulatory or permit decisions under the clean water act. In: Bruins RJF, Heberling MT, editors. Economics and ecological risk assessment: applications to watershed management. Boca Raton, FL: CRC Press; 2005. p. 401–21.
[72] Strange EM, Allen PD, Beltman D, Lipton J, Mills D. The habitat-based replacement cost method for assessing monetary damages for fish resource injuries. Fisheries 2004;29(7):17–24.
[73] Piehler C, de Mond J, Hamilton D, Finley H, Hanifen J, Lorentz W, et al. Damage assessment/restoration plan and environmental assessment: M/V Westchester crude oil discharge, lower Mississippi River, Louisiana, November 28, 2000. Final. Louisiana Oil Spill Coordinator's Office, Louisiana Department of Environmental Quality, Louisiana Department of Natural Resources, Louisiana Department of Wildlife and Fisheries; 2001.
[74] National Oceanic and Atmospheric Administration, Maryland Department of Natural Resources, Maryland Department of the Environment, U.S. Fish and Wildlife Service. Final restoration plan and environmental assessment for the April 7, 2000 Oil Spill at Chalk Point on the Patuxent River, Maryland. Silver Spring, Maryland: National Oceanic and Atmospheric Administration, Damage Assessment Center; 2002.
[75] Gittings SR, Bright TJ, Choi A, Barnett RR. The recovery process in a mechanically damaged coral reef community: recruitment and growth. Proceedings of the Sixth International Coral Reef Symposium, Australia, vol. 2; 1988. p. 226–30.
[76] Szmant AM. Nutrient effects on coral reefs: a hypothesis on the importance of topographic and trophic complexity to reef nutrient dynamics. Proceedings of the eighth international coral reef symposium, Panama, vol. 2; 1997. p. 1527–32.
[77] Miller MW, Barimo J. Assessment of juvenile coral populations at two reef restoration sites in the Florida Keys National Marine Sanctuary: indicators of success? Bulletin of Marine Science 2001;69(2):395–405.
[78] Rogers CS, Miller J. Permanent 'phase shifts' or reversible declines in coral cover? Lack of recovery of two coral reefs in St. John, US Virgin Islands. Marine Ecology Progress Series 2006;306:103–14.
[79] Hatcher BG. A maritime accident provides evidence for alternate stable states in benthic communities on coral reefs. Coral Reefs 1984;3:199–204.
[80] Knowlton N. Threshold and multiple stable states in coral reef community dynamics. American Zoologist 1992;32:674–82.
[81] Done TJ. Phase shifts in coral reef communities and their ecological significance. Hydrobiologia 1992;247:121–32.
[82] Precht WF, Aronson RB, Swanson DW. Improving scientific decision-making in the restoration of ship-grounding sites on coral reefs. Bulletin of Marine Science 2001;69(2):1001–12.
[83] Moulding AL. Coral recruitment patterns in the Florida Keys. Revista de Biologia Tropical 2005;53:75–82.
[84] Hughes TP, Tanner JE. Recruitment failure, life histories, and long-term decline of Caribbean corals. Ecology 2000;81(8):2250–63.
[85] Kojis BL, Quinn NJ. The importance of regional differences in hard coral recruitment rates for determining the need for coral restoration. Bulletin of Marine Science 2001;69(2):967–74.