# A comparative analysis of dynamic management in marine and terrestrial systems

**William K Oestreich[1*], Melissa S Chapman[2], and Larry B Crowder[1]**

**Protection of highly mobile species and shifting habitats is a practical challenge for conservation in both marine and terrestrial systems, particularly in light of the acceleration of land-use change and climate-driven range shifts. Static protected areas have long been a keystone of conservation but are generally insufficient for such species and habitats. Spatially and temporally dynamic management (DM) has arisen as a potential solution to this challenge. We present what we believe to be the first comparative analysis of DM across marine and terrestrial systems, focusing on the scales of DM approaches. Our results show that marine DM has largely been focused on relatively finer temporal scales, whereas terrestrial DM has focused on relatively finer spatial scales, often following the scale of available and relevant datasets. We explore not only the constraints imposed by data availability but also other drivers of DM trends and scales, highlighting areas in which exchange of approaches pioneered in each domain may be beneficial.**

***Front Ecol Environ*** 2020; doi:10.1002/fee.2243

Climate-driven range shifts and phenological changes in populations and habitats in both marine and terrestrial ecosystems are increasingly recognized as problems in applied ecology (Burrows *et al*. 2011). Many ecologists have recently highlighted – among other conservation and resource management concerns – the potential inadequacy of static protected areas both on land (Araújo *et al*. 2011; Scriven *et al*. 2015) and at sea (Hyrenbach *et al*. 2000; Hazen *et al*. 2013; Maxwell *et al*. 2020) in the face of such shifts. Further investigation reveals that the performance and efficacy of static approaches are vulnerable on shorter timescales to both climate-induced range shifts and the spatial dynamism associated with highly mobile species (either migratory or those with large home ranges; Taillon *et al*. 2012; Dunn *et al*. 2016) and seasonally shifting habitats (Reynolds *et al*. 2017). Combined with increasingly fine-scale data on environmental variables, wildlife movements, and resource-user behaviors, these findings have led to the rise of management strategies that use near-real-time data to dynamically delineate protected areas and mediate conflicts between conservation and other interests. Such dynamic management (DM) practices are now implemented or proposed for more effective management in a variety of contexts (Panel 1), including issues of land-use change (eg Figure 1b; Golet *et al*. 2018), human–wildlife conflict (eg Figure 1, a and c; O'Keefe and DeCelles 2013; Howell *et al*. 2015), and protection of vulnerable species under range shifts attributable to climate change or other disturbances (eg Figure 1d; Rayfield *et al*. 2008). The first generation of such DM programs has unfolded with both marked similarities and differences in marine and terrestrial systems. This approach has received substantial attention in oceanic systems, possibly because many marine ecosystems are more spatially dynamic than their terrestrial counterparts (Steele 1985), and it is often categorized under the umbrella term "dynamic ocean management" (DOM). Recent reviews on DOM have discussed its definition and conceptualization (Maxwell *et al*. 2015), key ingredients for its successful implementation (Lewison *et al*. 2015), and integration of the scientific and legal components of DOM (Hobday *et al*. 2013). On land, scholarship on these DM strategies has not coalesced under an umbrella concept comparable to DOM – an interesting difference between DM in marine and terrestrial systems in and of itself. For this reason, and for the sake of consistency, hereafter we refer to terrestrial examples of DM approaches as "dynamic terrestrial management" (DTM). Despite the lack of a widely used umbrella term, suggestions of DM practices on land (largely derived from agriculture and forestry) have a rich history predating the

## In a nutshell:

- Highly mobile species and shifting habitats have presented difficulties for traditional conservation measures, such as static protected areas
- Dynamic management (DM) approaches that "move" in space and time have arisen as potential supplements to static management both in the ocean and on land
- Our comparative analysis of DM in marine and terrestrial systems reveals that such programs often target different temporal and spatial scales
- We underscore opportunities for managers of marine systems to adopt DM approaches developed primarily for terrestrial settings, and vice versa

[1]*Hopkins Marine Station, Department of Biology, Stanford University, Pacific Grove, CA* [*]*(woestreich@stanford.edu);* [2]*Deparment of Environmental Science, Policy, and Management, University of California, Berkeley, Berkeley, CA*

**Panel 1. Diverse examples of proposed and implemented DM strategies in marine and terrestrial ecosystems**

**TurtleWatch (Figure 1a; Howell *et al*. 2015; 2018)**

This voluntary program aims to reduce bycatch of sea turtles in the Hawaii-based longline shallow-set fishery. Originally a single-species (loggerhead sea turtles, *Caretta caretta*) DM program, TurtleWatch has been expanded to include a second species (leatherback sea turtles, *Dermochelys coriacea*). This model allows managers to produce daily, 11-km resolution maps of areas to avoid fishing in order to reduce the likelihood of bycatch events. The model scale is primarily determined by the resolution of available satellite data and by the temporal scale relevant to resource users.

**BirdReturns (Figure 1b; Golet *et al*. 2018)**

This habitat-scale program is a voluntary reverse-auction-based initiative meant to provide temporary wetland habitat for migratory shorebirds in California's Sacramento Valley. Researchers with The Nature Conservancy (TNC) identify spatiotemporal gaps in migratory shorebird wetland habitat, convene workshops with local rice farmers to identify periods when alternative field flooding practices can be implemented, and run two reverse auctions annually in which farmers submit bids of acreage to be flooded as temporary migratory shorebird habitat in exchange for payment from TNC. The program scale is driven primarily by the size of resource-user bids and the temporal need for temporary habitat.

**Yellowtail flounder bycatch avoidance (Figure 1c; O'Keefe and DeCelles 2013)**

This program focuses on reducing bycatch of yellowtail flounder (*Limanda ferruginea*) in the US Atlantic sea scallop (*Placopecten magellanicus*) fishery. Unlike many other DM programs focused on mitigating bycatch, this initiative utilizes near-real-time bycatch data from fishermen to create a daily risk grid of bycatch potential. This approach, while not necessarily predictive, circumvents the need for gridded environmental data products and their associated limitations on spatial and temporal scale. Instead, the scale of this program is primarily driven by the suggestions of stakeholders.

**Boreal forest DM for protection of American marten (Figure 1d; Rayfield *et al*. 2008)**

This proposed DM program seeks to delineate dynamic protected areas for American marten (*Martes americana*) within the multi-use, fire-vulnerable boreal forest landscape of Quebec, Canada. The modeling effort aims to identify the arrangement of protected areas that minimize cost (a function of marten habitat density, boundary lengths, and penalties associated with not reaching targets for total protected area). The number of dynamic protected areas (two) and the temporal scale of protected area shifts (50 years) are primarily driven by the operational feasibility of dynamic protected areas and the costs associated with their movement.

development of the ecological modeling tools often necessary for implementing such approaches (Gustafson 1996). DTM initiatives, along with their underlying near-term ecological forecasting and dynamic species distribution models, have received increased attention as a topic of research in recent years (Chetkiewicz and Boyce 2009; Franklin 2010; Dietze *et al*. 2018), largely spurred by the intensity of global land-use change and increasingly apparent climate-driven range shifts of terrestrial species (Bonebrake *et al*. 2018).

Effective implementation of DM strategies will require not only integration of several underlying social, ecological, and governance components and processes (Figure 2; Lewison *et al*. 2015) with ecological forecasts (predictions of future conditions), nowcasts (predictions of present conditions), and species distribution models, but also stakeholder engagement (Guisan *et al*. 2013). Nonetheless, ecological modeling and the scales at which these models can be developed and implemented remain central issues in the field of DM. Scale in particular, in terms of time, space, and the number of species and interactions considered, is central to both ecological modeling and DM objectives (Lewison *et al*. 2015). Choice of scale can have profound impacts on model accuracy and uncertainty (Araújo *et al*. 2005), and inherently determines the social, ecological, and governance processes that can be addressed via dynamic approaches. The issue of scale is so central to dynamic conservation decision making that optimal decision-making strategies can be greatly affected by the scale of species distribution models alone (Franklin *et al*. 2013).

Here, we present – to the best of our knowledge – the first comparative analysis of DM in marine and terrestrial systems. We identify current strengths and limitations on the scale of the ecological models underlying DOM and DTM strategies (Figure 2). We then discuss the drivers of disparities between DOM and DTM scales (Figure 4) and opportunities for closing those gaps in the future. Despite differences in the application of ecological modeling to DOM and DTM, we argue that approaches primarily used in the marine domain will benefit DM in the terrestrial domain, and vice versa.

## Methods

To investigate the scales of existing ecological models for DM practices, we conducted a systematic literature review via Google Scholar searches. The goal of this systematic review was to identify ecological models and observation systems explicitly developed, at least in part, for the purpose of spatiotemporally DM strategies. We included all cases describing both an ecological model or near-real-time datastream and its role in implemented, proposed, or hypothetical DM schemes. Descriptions of "standardized time–area closures", or protected areas that shift in space and time but on a trajectory not updated in a dynamic manner using near-real-time data sources, were excluded from the analysis. Cases


NOAA T Koerner NOAA JW Frank

**Figure 1.** Example species and socioecological systems targeted by proposed and implemented dynamic management (DM) strategies: (a) loggerhead sea turtle (*Caretta caretta*) bycatch avoidance (TurtleWatch; Howell *et al*. 2008, 2015); (b) habitat provisioning for migratory shorebirds, including Wilson's phalarope (*Phalaropus tricolor*; BirdReturns; Golet *et al*. 2018); (c) yellowtail flounder (*Limanda ferruginea*) bycatch avoidance in the US Atlantic sea scallop (*Placopecten magellanicus*) fishery (O'Keefe and DeCelles 2013); and (d) protected areas for American marten (*Martes americana*) in forested landscapes of Quebec, Canada (Rayfield *et al*. 2008). See Panel 1 for more detailed descriptions.

describing analyses, tools, or programs potentially relevant to DM practices but that were not explicitly discussed in a DM context by the study authors were also excluded. Searches targeting both marine and terrestrial domains were conducted (Table 1), yielding a total of 1078 total search results (as of 30 Jan 2019). Each of the search results, as well as any relevant references therein, were considered for inclusion in our analysis. The use of broad search terms was intentional, owing to the diverse and wide-ranging types of studies describing DM programs; however, given the wide range of studies detected by our broad search terms, the vast majority of these 1078 search results were not relevant to the present study. We refined our wide-ranging initial search to a final list of case studies by first scanning search result abstracts for relevance, then reading papers identified as relevant for final determination of inclusion in this meta-analysis, and finally searching references therein for any additional case studies meeting the aforementioned criteria.

Several attributes for all cases meeting the criteria for inclusion were recorded; these consisted of the domain (marine or terrestrial), the temporal and spatial scales of the model or program, the number of species included, factors determining the temporal and spatial scales of the model or program, factors limiting implementation or success of the model or program, and relevant metadata (WebTable 1). All temporal and spatial scales recorded for these cases and discussed throughout the text refer to the "grain" or "resolution" element of scale, rather than the "extent" (Turner *et al*. 2001).

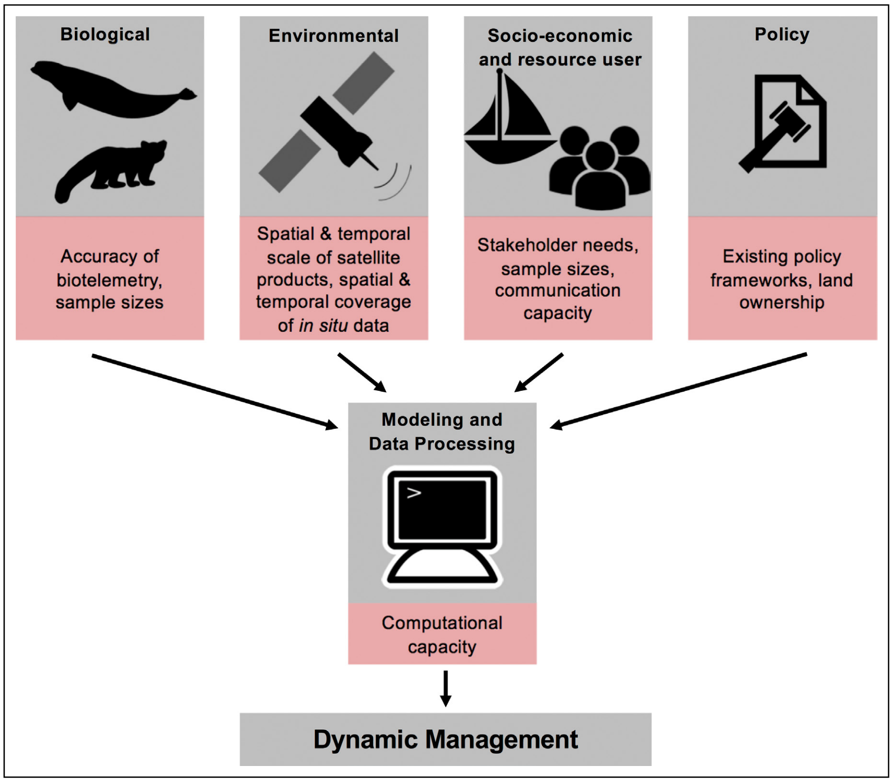


**Figure 2.** Conceptual diagram of the key components of DM (Hobday *et al.* 2013; Lewison *et al.* 2015; Maxwell *et al.* 2015) and associated potential limitations (in red) on program scale. Note that not all programs necessarily require all elements shown here, although most require some combination of these key components.

Unit conversions were necessary to allow for direct comparison of scales (in units of days, kilometers, and number of species; see WebPanel 1 for details). Although all conversions were calculated with the greatest accuracy possible based on information contained in each case study's reference(s), some reported values (WebTable 1) were approximated due to a range of or ambiguity in reported scales. Cases described as "near-real-time" by the authors were recorded as such and are presented approximately in comparisons of DM scales. Most identified cases described "nowcast" models updated at a clearly defined temporal scale, but in a few rare cases (eg Spillman *et al.* 2015) forecasting models included two temporal scales: the temporal scale of forecasting model updates, and the "lead time" with which the forecast aimed to project ecological dynamics. In such cases, only the temporal scale of model updates was recorded for the sake of comparability to the other identified case studies.

**Table 1. Google Scholar search terms and results used for identification of case studies**

| Search terms | Target domain | Number of results |
|---|---|---|
| "dynamic ocean management" | Marine | 313 |
| "dynamic management", "migratory", "land" | Terrestrial | 396 |
| "dynamic spatial management" | Both | 185 |
| "spatial management", "habitat predictions" | Both | 184 |

**Notes:** number of results reported for date of search (30 Jan 2019).

## Results

We identified 46 cases matching the criteria described in the Methods section above (see also WebTable 1), of which 34 (74%) focused on marine systems and 12 (26%) on terrestrial systems. Forty-three of these cases were either published or first implemented in the years 2007–2019, with only three case studies predating 2007 (Figure 3). For all models identified in the case studies, the spatial scale ranged from 0.005 to 761 km (Figure 4; WebTable 1), and the temporal scale ranged from 0.04 to 18,262 days (Figure 4; WebTable 1). The number of species included in models from these 46 case studies ranged from 1 to 57, but the majority (27 cases) considered a single species (Figure 5; WebTable 1). Four case studies described models at the habitat scale rather than for individual species, and one case study described models for a large number of species comprising a single clade of organisms (birds; Van Doren and Horton 2018). Of the identified cases, two described implemented and regulatory management programs, 12 described implemented but voluntary management programs, and 32 described proposed management programs. Concerning factors driving the scale of DM, the majority (27 of 46) of case studies cited the scale of available environmental data or associated factors (eg matching cell sizes for grids of spatial environmental datasets [such as satellite data products], temporal averaging of satellite data to avoid cloud-cover issues) as a factor determining program or model scale. Other factors cited by several case studies included the scale of ecological processes relevant to the study organism or system and stakeholder needs and suggestions. Regarding limitations to the implementation or success of DM, many case studies did not explicitly discuss these impediments, perhaps because most cases were merely proposed as opposed to actually implemented. Of those cases that were implemented, legal and policy constraints and costs were most often cited as limiting factors to DM implementation and success.

## Discussion

### Timeline of publications on DM

The timing of publication and implementation of ecological models for DM confirms that this is a relatively nascent field of research, with the majority of relevant studies published after 2007 (Figure 3). The rise in ecological models for DM has accelerated since 2011, with at least one such case study focusing on marine systems published each year over the period from 2011 to 2019. The growing number of such publications over the past decade or so, particularly for marine systems, coincides with the timing of key conceptual pieces on this topic (Lewison *et al.* 2015; Maxwell *et al.* 2015; Bonebrake *et al.* 2018). However, several notable outliers to this trend should be mentioned. The first outlier, chronologically speaking, is the US Department of Agriculture's Conservation Reserve Program (CRP; Glaser 1986), which was initially implemented as a part of the Food Security Act of 1985 (Public Law 99-198). This federal program (along with subsequent state-level subprograms) provides income to farmers in exchange for acreage deemed suitable to be set aside over decade-long leases for one of several conservation purposes, including "creat[ing] better habitat for fish and wildlife through improved food and cover" (Federal Register 1987; **52** [28]: 4269; https://bit.ly/37B7HVZ). Another outlier is BushTender, a program similar to the CRP overseen by Australia's Victorian Department of Natural Resources (first implemented in 2000; Stoneham *et al.* 2003). Although the time horizon of such lease programs (annual to decadal) and the spatial modeling element of the CRP and BushTender are far less dynamic and analytically advanced as compared to more recent applications of DTM strategies, these programs do take a spatiotemporally dynamic approach to balancing the economic interests of farmers with biodiversity and other environmental goals. The conceptual basis of these terrestrial programs – mitigation of issues related to land-use change via habitat-scale modeling at relatively long timescales – is reflected decades later in more recent DTM approaches that involve much more sophisticated underlying ecological modeling, and may be a relevant and valuable conceptual approach for future DOM initiatives. The first DOM case study identified in this review – that of halibut and crab bycatch avoidance in the Bering Sea trawl fishery (Gauvin *et al.* 1995) – predates all other identified DOM programs by more than a decade. Despite the absence of the technological and modeling advances that facilitated later DOM programs, the Bering Sea trawl fishery program was made possible through the use of near-real-time information collected by resource users themselves, an approach (ie user-reported spatial bycatch data) and application (ie to resolve fisheries bycatch) that have since been adopted by other DOM programs (eg O'Keefe and DeCelles 2013). In more recent years, however, proposals and adoptions of DOM programs have also been applied to a marked number and variety of marine resource management issues, as reflected by the majority of DM case studies to date originating from marine systems (Figure 3). Similarly, greater adoption of DM approaches in the terrestrial realm may be beneficial for a wide variety of management issues on land.

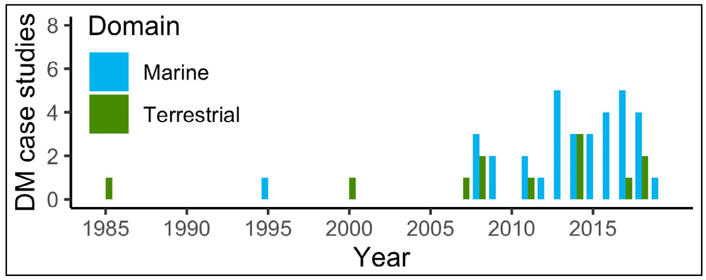


**Figure 3.** Number of identified case studies, sorted by year of publication, from marine and terrestrial systems reporting on implemented, proposed, or hypothetical DM strategies.

## Scales of DM

In marine and terrestrial systems, models are typically produced for DM at different spatial and temporal scales (Figures 4 and 5). The most prominent trends in scale that we identified are as follows: (1) DOM cases tend to cover relatively coarser spatial scales but over relatively finer temporal scales, whereas DTM cases tend to cover a broader range of spatial scales but over relatively longer temporal scales (Figure 4); (2) the combination of the trends identified in (1) yields a considerable gap in DTM and DOM models at both fine temporal and spatial scales (Figure 4); (3) the majority of DM models consider a single species, although a higher proportion of DTM cases (roughly half) model at

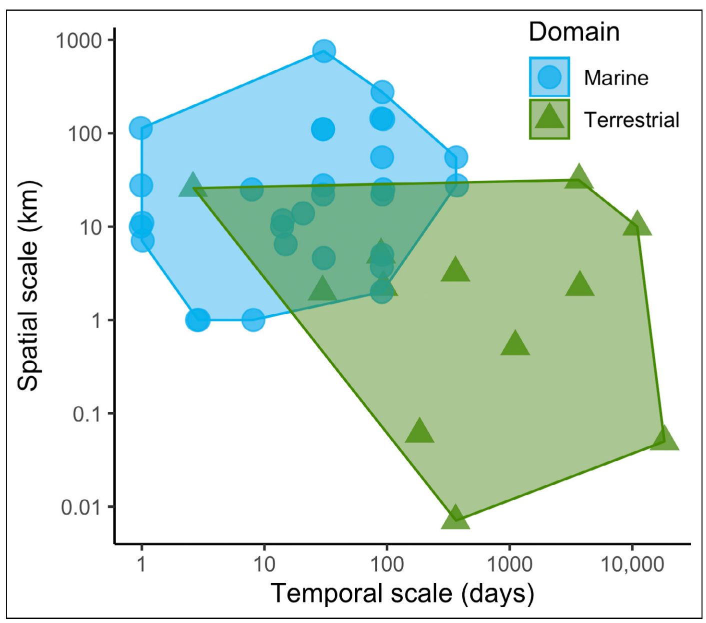


**Figure 4.** Spatial and temporal scales of DM case studies identified in this analysis. Scales have been converted and normalized as described in the Methods section. Circles and triangles represent case studies in marine and terrestrial domains, respectively. Overlapping data points are slightly shifted for display purposes and to avoid masking of multiple data points at the same spatial and temporal scales. Note the logarithmic scale of axes. Only cases with both clearly defined spatial and temporal scales shown here.

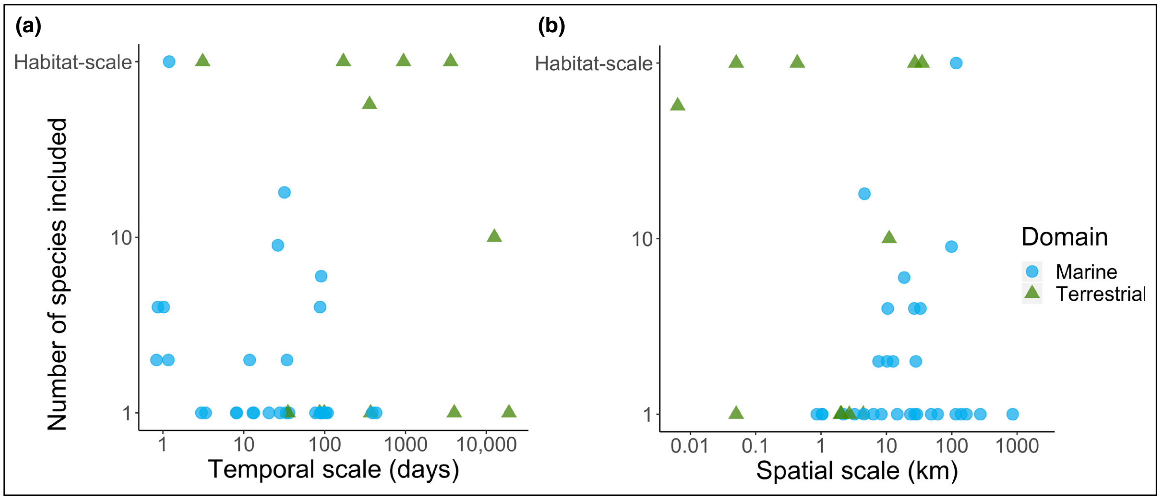


**Figure 5.** Comparison of number of species considered versus (a) temporal scale and (b) spatial scale for DM case studies identified in this analysis. Scales have been converted and normalized as described in the Methods section. Circles and triangles represent case studies in marine and terrestrial domains, respectively. Overlapping data points are slightly shifted for display purposes and to avoid masking of multiple data points at the same spatial and temporal scales. Note the logarithmic scale of axes, as well as the designation of "habitat scale" as the maximum y axis value (for more details, see WebPanel 1). Only cases with both clearly defined spatial and temporal scales shown here.

the habitat scale as compared to DOM cases (Figure 5); and finally, (4) there are no clear trade-offs between the number of species modeled and model spatial or temporal scales (Figure 5).

These gaps and trends may be attributable to several factors, including: the scales of ecological processes and resource management issues perceived to be potential candidates for DM in marine and terrestrial systems; the limitations on scale imposed via current data availability; and constraints on scale or implementation as a result of other components of the DM process, including governance, policy design, and stakeholder engagement (Figure 2). In the remainder of this discussion, we highlight how these factors both determine and limit the appropriate scales of modeling for DM, and how recent developments may be leveraged to close these gaps moving forward.

## Constraints on the scale of DM

### *Scales of relevant ecological processes and resource management issues*

Marine and terrestrial systems face resource management and conservation challenges across the range of spatial and temporal scales explored here, but the trends we identified suggest that DOM and DTM have – to date – been developed to address challenges at different scales. The coarser temporal scale and relatively larger proportion of high species number and habitat-scale DTM cases suggest that DTM is currently leveraged as a tool to address longer temporal scale concerns (eg climate-induced range shifts, seasonal migrations; Bull *et al.* 2013; Alagador *et al.* 2014) for resource management questions at the multispecies or habitat level (eg land-use change and management; Golet *et al.* 2018). In contrast, the finer temporal scale and relatively smaller proportion of high species number and habitat-scale DOM cases suggest that DOM is used and proposed more as a tool to address shorter temporal scale concerns (eg sensitive species' movements at daily, seasonal, and interannual scales in relation to fixed and dynamic oceanographic drivers; Hobday *et al.* 2011) for resource management questions relevant to a single or a handful of species of interest (eg bycatch and other human–wildlife conflicts; Hazen *et al.* 2018).

These trends may reflect the relative scales of ecological processes underlying species movements in terrestrial and marine systems. The inherent spatiotemporal dynamism of primary productivity and associated consumers in marine ecosystems (Steele 1985; Benoit-Bird *et al.* 2013; Kavanaugh *et al.* 2016) makes fine temporal scales a natural starting point for application of DOM. However, marine resource management issues of coarser temporal dynamism akin to terrestrial climate-change and land-use-change issues (eg coastal and pelagic aquaculture "sea-use" operations; Spillman *et al.* 2015) may be candidates for DOM at coarser temporal scales. Furthermore, underlying climate-induced range shifts in marine ecosystems are critical to consider in coarser temporal scale DOM programs alongside the influence of shorter term oceanographic drivers (Hazen *et al.* 2013). For terrestrial systems, in contrast, the application of DM to longer term landscape-scale resource management concerns (Glaser 1986) is a natural starting point for DTM given the relatively fixed nature of many terrestrial ecosystems on finer timescales. Of course, some terrestrial ecosystems (eg wetlands) are dramatically spatiotemporally dynamic on seasonal to annual scales, making them prime candidates for somewhat finer temporal-scale DM (Panel 1; Golet *et al.* 2018). Yet even finer temporal-scale ecological modeling and DTM might benefit terrestrial resource management. As shown in DOM applications, the fine-scale movements of sensitive species and their interactions with human resource users may be modeled on relatively fine temporal scales in order to mitigate acute human–wildlife conflict issues in terrestrial systems with spatially dynamic wildlife and human land uses (Mueller *et al.* 2008). For example, dynamic road closures on seasonal or finer temporal scales (Whittington *et al.* 2019) based on modeled movements of sensitive species may be an achievable and effective conservation measure in terrestrial ecosystems.

The general trend of shorter temporal and coarser spatial scales for DOM as opposed to longer temporal and finer spatial scales for DTM is a natural starting point given known trends in ecological scale in these ecosystem types, and may explain why there has so far been limited exchange between marine and terrestrial systems in the types of resource management addressed via DM. Indeed, for several case studies that we identified, the scales most ecologically relevant to the species or resource of interest are explicitly stated as a driver of the program scales (WebTable 1). However, the vast majority of identified case studies cite alternative reasons for the chosen spatial and temporal scales, suggesting that the scale of ecological processes does not exclusively determine the scale of DM programs.

### *Data availability*

The resource management issues that could be addressed with DM are also inherently limited by the scales of available underlying data and model methodologies; indeed, most of the case studies included here cite scales of available relevant data and associated concerns (eg matching grids of environmental data products, binning or averaging to overcome data gaps) as a driver of program scale. Earth-observing satellite data in particular are a common choice for environmental information in DM modeling, as evidenced by many programs operating at the scales of well-known global satellite data products (Figure 4). This does not necessarily indicate that such DM programs are developed at scales that differ from the most relevant ecological or resource management dynamics, as the scale of satellite observation programs is chosen in part to match the scale of the ecological processes they aim to observe (Kerr and Ostrovsky 2003). However, two pieces of evidence suggest that DM programs can and do extend beyond the scales of currently available satellite-based remote-sensing data. First, from a theoretical standpoint, one would expect processes at multiple spatial and temporal scales to affect the emergent ecological dynamics of a species or habitat of interest (ie the influence of processes at multiple scales should be considered in DM applications; Scales *et al.* 2017; Hazen *et al.* 2018). Second, from an empirical perspective, several case studies that we identified use alternative data sources and modeling methods that circumvent the need for remote-sensing data (Panel 1; eg O'Keefe and DeCelles 2013), and operate on fundamentally different scales (eg Wiley *et al.* 2013). Despite being the most commonly cited scale limiter by the case studies identified here, environmental data sources are not the only data- and methodology-related constraints on DM program scale (Figure 2). For a more comprehensive assessment of how other data sources, modeling methods, and uncertainty can determine DM scales, see WebPanel 2.

### *Governance, policy design, and stakeholder engagement*

In addition to the analytical tools that we primarily focus on here, stakeholder engagement, incentive structures, policy design, dissemination of information, and cost analysis have also been identified as critical components of effective DM (Figure 2; Alagador *et al.* 2014; Lewison *et al.* 2015; Welch *et al.* 2019). As the analytical tools required for DM modeling across scales continue to be developed (WebPanel 2), these additional elements of DM must also be developed to allow for effective application of DM at these scales. For example, as models of increasing statistical complexity enable multispecies DM or reduction of uncertainty across temporal scales, language and strategies for communicating such models to managers and stakeholders must be considered (Cartwright *et al.* 2016). Sustained interaction with stakeholders – along with knowledge exchange – throughout the data collection and model development stages of DM programs may be critical for increased uptake of proposed DM programs as well, especially at finer spatial and temporal scales (Danielsen *et al.* 2010; Cvitanovic *et al.* 2015). The case studies reviewed in our analysis further indicate the importance of stakeholder engagement and communication as keys to DM implementation and success, with several cases making explicit reference to such concerns in discussion of program implementation (WebTable 1).

Existing policy frameworks (eg land ownership in perpetuity [Greene 2004]; limited marine application of the Public Trust Doctrine, a natural resource management trust applied to only state and not federal waters [Turnipseed *et al.* 2009]; international waters beyond national jurisdiction [Maxwell *et al.* 2020]) that limit situations in which habitats and species can be dynamically managed must be addressed as well, in order to facilitate the launch of new DM programs. Legal scholars have highlighted even higher-level policy barriers to dynamic conservation approaches, including the need to recognize spatial dynamism and multiscale processes in conservation law (McDonald *et al.* 2016). These paradigms, as well as the difficulty of establishing multinational agreements to carry out DM for species crossing international boundaries, are regularly cited in the identified case studies as barriers to program implementation (in the case of proposed DM) and success (in the case of implemented DM) (WebTable 1). "No net loss" biodiversity offset schemes (Bull *et al.* 2013), which are used primarily in terrestrial systems, can provide a policy mechanism for DM and could expand the range of viable DOM scenarios if adopted in marine systems. Alternatively, some non-regulatory DOM (Panel 1; Howell *et al.*

2015) and DTM (Panel 1; Golet *et al*. 2018) approaches have circumvented the need for DM-friendly policy design via voluntary or incentive-based programs. Given that spatiotemporal dynamism in legal and policy frameworks is often difficult to enact (Greene 2004), such non-regulatory approaches may expand the possibility of DM application at scales for which dynamic policy designs are unlikely or impractical.

## Conclusions

In this review, we demonstrated that the first generation of ecological models for DM have primarily been implemented to address resource management issues at different scales in terrestrial and marine systems. These current gaps in DM application may be closed by increased exchange and sharing of analytical tools, data-gathering approaches, and ways of thinking about the utility of DM across terrestrial and marine systems. In terrestrial systems, application of DM thinking and ecological modeling to a variety of conservation and resource management issues may lead to broader uptake of this approach similar to that for marine systems. Likewise, in marine systems, increased application of DM at the seascape level will be valuable. The analyses described here suggest that use of diverse environmental, wildlife, and resource-user data sources in particular could aid in expanding the range of scales to which DM is applied across both system types. Undoubtedly, the lessons learned from DM to date are limited by the relatively small number of proposed, and even fewer implemented, DM programs. We encourage broader application of and experimentation with DM thinking in diverse resource management contexts in order to improve understanding of the utility of DM. In addition, increased information exchange and comparison of DM approaches across the marine and terrestrial domains are critical to provide managers with the next generation of dynamic tools needed to address current and future conservation and resource management challenges.

## Acknowledgements

We thank B Abrahms and H Welch of the NOAA Southwest Fisheries Science Center for helpful discussions on contextualizing the findings of the meta-analysis presented here. WKO is supported by the US National Science Foundation (NSF) Graduate Research Fellowship Program and as the David and Lucile Packard Foundation Stanford Graduate Fellow. MSC is supported by NSF grant #DGE-1450053. All data and code for the analyses and figures presented here available at: https://github.com/milliechapman/dynamic-management-review.

## References

Alagador D, Cerdeira JO, and Araújo MB. 2014. Shifting protected areas: scheduling spatial priorities under climate change. *J Appl Ecol* **51**: 703–13.

Araújo MB, Alagador D, Cabeza M, *et al*. 2011. Climate change threatens European conservation areas. *Ecol Lett* **14**: 484–92.

Araújo MB, Thuiller W, Williams PH, and Reginster I. 2005. Downscaling European species atlas distributions to a finer resolution: implications for conservation planning. *Global Ecol Biogeogr* **14**: 17–30.

Benoit-Bird KJ, Battaile BC, Heppell SA, *et al*. 2013. Prey patch patterns predict habitat use by top marine predators with diverse foraging strategies. *PLoS ONE* **8**: e53348.

Bonebrake TC, Brown CJ, Bell JD, *et al*. 2018. Managing consequences of climate-driven species redistribution requires integration of ecology, conservation and social science. *Biol Rev* **93**: 284–305.

Bull JW, Suttle KB, Singh NJ, and Milner-Gulland EJ. 2013. Conservation when nothing stands still: moving targets and biodiversity offsets. *Front Ecol Environ* **11**: 203–10.

Burrows MT, Schoeman DS, Buckley LB, *et al*. 2011. The pace of shifting climate in marine and terrestrial ecosystems. *Science* **334**: 652–55.

Cartwright SJ, Bowgen KM, Collop C, *et al*. 2016. Communicating complex ecological models to non-scientist end users. *Ecol Model* **338**: 51–59.

Chetkiewicz CB and Boyce MS. 2009. Use of resource selection functions to identify conservation corridors. *J Appl Ecol* **46**: 1036–47.

Cvitanovic C, Hobday AJ, van Kerkhoff L, *et al*. 2015. Improving knowledge exchange among scientists and decision-makers to facilitate the adaptive governance of marine resources: a review of knowledge and research needs. *Ocean Coast Manage* **112**: 25–35.

Danielsen F, Burgess ND, Jensen PM, and Pirhofer-Walzl K. 2010. Environmental monitoring: the scale and speed of implementation varies according to the degree of peoples involvement. *J Appl Ecol* **47**: 1166–68.

Dietze MC, Fox A, Beck-Johnson LM, *et al*. 2018. Iterative near-term ecological forecasting: needs, opportunities, and challenges. *P Natl Acad Sci USA* **115**: 1424–32.

Dunn DC, Maxwell SM, Boustany AM, and Halpin PN. 2016. Dynamic ocean management increases the efficiency and efficacy of fisheries management. *P Natl Acad Sci USA* **113**: 668–73.

Franklin J. 2010. Moving beyond static species distribution models in support of conservation biogeography. *Divers Distrib* **16**: 321–30.

Franklin J, Davis FW, Ikegami M, *et al*. 2013. Modeling plant species distributions under future climates: how fine scale do climate projections need to be? *Glob Change Biol* **19**: 473–83.

Gauvin JR, Haflinger K, Nerini M, and Wray T. 1995. Implementation of a voluntary bycatch avoidance program in the flatfish fisheries of the eastern Bering Sea. In: Solving bycatch: considerations for today and tomorrow. Fairbanks, AK: University of Alaska.

Glaser LK. 1986. Provisions of the Food Security Act of 1985. Washington, DC: US Department of Agriculture.

Golet GH, Low C, Avery S, *et al*. 2018. Using ricelands to provide temporary shorebird habitat during migration. *Ecol Appl* **28**: 409–26.

Greene DM. 2004. Dynamic conservation easements: facing the problem of perpetuity in land conservation. *Seattle U Law Rev* **28**: 883–923.

Guisan A, Tingley R, Baumgartner JB, *et al*. 2013. Predicting species distributions for conservation decisions. *Ecol Lett* **16**: 1424–35.

Gustafson EJ. 1996. Expanding the scale of forest management: allocating timber harvests in time and space. *Forest Ecol Manag* **87**: 27–39.

Hazen EL, Jorgensen S, Rykaczewski RR, *et al*. 2013. Predicted habitat shifts of Pacific top predators in a changing climate. *Nat Clim Change* **3**: 234.

Hazen EL, Scales KL, Maxwell SM, *et al*. 2018. A dynamic ocean management tool to reduce bycatch and support sustainable fisheries. *Sci Adv* **4**: eaar3001.

Hobday AJ, Hartog JR, Spillman CM, and Alves O. 2011. Seasonal forecasting of tuna habitat for dynamic spatial management. *Can J Fish Aquat Sci* **68**: 898–911.

Hobday AJ, Maxwell SM, Forgie J, and McDonald J. 2013. Dynamic ocean management: integrating scientific and technological capacity with law, policy, and management. *Stanford Environ Law J* **33**: 125–65.

Howell EA, Hoover A, Benson SR, *et al*. 2015. Enhancing the TurtleWatch product for leatherback sea turtles, a dynamic habitat model for ecosystem-based management. *Fish Oceanogr* **24**: 57–68.

Howell EA, Kobayashi DR, Parker DM, *et al*. 2008. TurtleWatch: a tool to aid in the bycatch reduction of loggerhead turtles *Caretta caretta* in the Hawaii-based pelagic longline fishery. *Endanger Species Res* **5**: 267–78.

Hyrenbach KD, Forney KA, and Dayton PK. 2000. Marine protected areas and ocean basin management. *Aquat Conserv* **10**: 437–58.

Kavanaugh MT, Oliver MJ, Chavez FP, *et al*. 2016. Seascapes as a new vernacular for pelagic ocean monitoring, management and conservation. *ICES J Mar Sci* **73**: 1839–50.

Kerr JT and Ostrovsky M. 2003. From space to species: ecological applications for remote sensing. *Trends Ecol Evol* **18**: 299–305.

Lewison R, Hobday AJ, Maxwell S, *et al*. 2015. Dynamic ocean management: identifying the critical ingredients of dynamic approaches to ocean resource management. *BioScience* **65**: 486–98.

Maxwell SM, Gjerde KM, Conners MG, and Crowder LB. 2020. Mobile protected areas for biodiversity on the high seas. *Science* **367**: 252–54.

Maxwell SM, Hazen EL, Lewison RL, *et al*. 2015. Dynamic ocean management: defining and conceptualizing real-time management of the ocean. *Mar Policy* **58**: 42–50.

McDonald J, McCormack PC, and Foerster A. 2016. Promoting resilience to climate change in Australian conservation law: the case of biodiversity offsets. *U New South Wales Law J* **39**: 1612.

Mueller T, Olson KA, Fuller TK, *et al*. 2008. In search of forage: predicting dynamic habitats of Mongolian gazelles using satellite-based estimates of vegetation productivity. *J Appl Ecol* **45**: 649–58.

O'Keefe CE and DeCelles GR. 2013. Forming a partnership to avoid bycatch. *Fisheries* **38**: 434–44.

Rayfield B, James PM, Fall A, and Fortin M-J. 2008. Comparing static versus dynamic protected areas in the Quebec boreal forest. *Biol Conserv* **141**: 438–49.

Reynolds MD, Sullivan BL, Hallstein E, *et al*. 2017. Dynamic conservation for migratory species. *Sci Adv* **3**: e1700707.

Scales KL, Hazen EL, Jacox MG, *et al*. 2017. Scale of inference: on the sensitivity of habitat models for wide-ranging marine predators to the resolution of environmental data. *Ecography* **40**: 210–20.

Scriven SA, Hodgson JA, McClean CJ, and Hill JK. 2015. Protected areas in Borneo may fail to conserve tropical forest biodiversity under climate change. *Biol Conserv* **184**: 414–23.

Spillman CM, Hartog JR, Hobday AJ, and Hudson D. 2015. Predicting environmental drivers for prawn aquaculture production to aid improved farm management. *Aquaculture* **447**: 56–65.

Steele JH. 1985. A comparison of terrestrial and marine ecological systems. *Nature* **313**: 355–58.

Stoneham G, Chaudhri V, Ha A, and Strappazzon L. 2003. Auctions for conservation contracts: an empirical examination of Victoria's BushTender trial. *Aust J Agr Resour Ec* **47**: 477–500.

Taillon J, Festa-Bianchet M, and Côté SD. 2012. Shifting targets in the tundra: protection of migratory caribou calving grounds must account for spatial changes over time. *Biol Conserv* **147**: 163–73.

Turner MG, Gardner RH, and O'Neill RV. 2001. Landscape ecology in theory and practice. New York, NY: Springer.

Turnipseed M, Crowder LB, Sagarin RD, and Roady SE. 2009. Legal bedrock for rebuilding America's ocean ecosystems. *Science* **324**: 183–84.

Van Doren BM and Horton KG. 2018. A continental system for forecasting bird migration. *Science* **361**: 1115–18.

Welch H, Hazen EL, Bograd SJ, *et al*. 2019. Practical considerations for operationalizing dynamic management tools. *J Appl Ecol* **56**: 459–69.

Whittington J, Low P, and Hunt B. 2019. Temporal road closures improve habitat quality for wildlife. *Sci Rep-UK* **9**: 3772.

Wiley D, Hatch L, Schwehr K, *et al*. 2013. Marine sanctuaries and marine planning. *Coast Guard Proc Mar Safety Security Council* **70**: 10–15.



## Supporting Information

Additional, web-only material may be found in the online version of this article at http://onlinelibrary.wiley.com/doi/10.1002/fee.2243/suppinfo