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Rapidly Changing Range Limits in a Warming World: Critical Data Limitations and Knowledge Gaps for Advancing Understanding of Mangrove Range Dynamics in the Southeastern USA

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Received: 8 January 2023 / Revised: 20 March 2023 / Accepted: 12 April 2023 / Published online: 9 May 2023
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Abstract

Climate change is altering species' range limits and transforming ecosystems. For example, warming temperatures are leading to the range expansion of tropical, cold-sensitive species at the expense of their cold-tolerant counterparts. In some temperate and subtropical coastal wetlands, warming winters are enabling mangrove forest encroachment into salt marsh, which is a major regime shift that has significant ecological and societal ramifications. Here, we synthesized existing data and expert knowledge to assess the distribution of mangroves near rapidly changing range limits in the southeastern USA. We used expert elicitation to identify data limitations and highlight knowledge gaps for advancing understanding of past, current, and future range dynamics. Mangroves near poleward range limits are often shorter, wider, and more shrublike compared to their tropical counterparts that grow as tall forests in freeze-free, resource-rich environments. The northern range limits of mangroves in the southeastern USA are particularly dynamic and climate sensitive due to abundance of suitable coastal wetland habitat and the exposure of mangroves to winter temperature extremes that are much colder than comparable range limits on other continents. Thus, there is need for methodological refinements and improved spatiotemporal data regarding changes in mangrove structure and abundance near northern range limits in the southeastern USA. Advancing understanding of rapidly changing range limits is critical for foundation plant species such as mangroves, as it provides a basis for anticipating and preparing for the cascading effects of climate-induced species redistribution on ecosystems and the human communities that depend on their ecosystem services.

Keywords Climate change · Coastal wetlands · Expert elicitation · Range limit · Range expansion · Mangrove distribution

Communicated by Kenneth L. Heck

Rémi Bardou, Michael J. Osland, and A. Randall Hughes shared co-first authorship. These authors contributed equally to this publication and share first authorship.

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Introduction

Climate change is transforming ecosystems and leading to the global redistribution of species (Pecl et al. 2017; USGCRP 2018; IPCC 2022). In North America, within the transition zone between tropical and temperate climates, cold temperature extremes in winter govern the poleward range limits of many tropical, cold-sensitive species (Sakai and Larcher 1987;

Boucek et al. 2016; Osland et al. 2021). In response to warming winters, many of these tropical species are expected to move northward, sometimes replacing their temperate counterparts. Changing winter temperature regimes can blur the boundaries between formerly distinct ecological communities, producing novel tropical-temperate transition zones where new combinations of species overlap (D’Odorico et al. 2013; Vergés et al. 2014; Carter et al. 2018). Ecologists and natural resource managers working within these transition zones are increasingly challenged to better anticipate and prepare for the ecological impacts of warming-induced range expansions (Pinsky et al. 2018; Henry and Sorte 2021). Thus, there is a pressing need for information regarding past and present range dynamics within these transition zones (Osland et al. 2021; Walters and McClenachan 2021; Ximenes et al. 2022), which is critical information for projecting and preparing for future range expansions and ecological regime shifts due to climate change.

This communication focuses on coastal wetlands in the southeastern USA, where warming winter temperatures are enabling the poleward range expansion of tropical and subtropical, cold-sensitive mangrove forests, sometimes at the expense of cold-tolerant salt marsh plants (Gabler et al. 2017; Cavanaugh et al. 2019). Mangrove and salt marsh plants are both considered foundation plant species that create habitat, modulate ecosystem dynamics, and facilitate the development of entire ecological communities (Dayton 1972; Ellison et al. 2005; Ellison 2019). However, mangrove range expansion can affect the critical ecosystem goods and services provided by coastal wetlands—for example, coastal protection from storms, coastal fisheries, carbon sequestration, avian habitat, and recreational opportunities. Mangrove expansion and marsh displacement can produce favorable changes in some ecosystem services, while simultaneously producing unfavorable changes in other services (Kelleway et al. 2017; Osland et al. 2022).

Mangrove expansion is a global phenomenon that is occurring on many continents in response to various global change drivers (Saintilan et al. 2014; Saintilan and Rogers 2015; Rogers and Krauss 2019), but the northern range limits in the southeastern USA are particularly dynamic and climate sensitive due to (1) abundance of suitable coastal wetland habitat near and beyond mangrove range limits (Stedman and Dahl 2008); and (2) the exposure of mangroves to winter temperature extremes that are much colder than comparable range limits in other regions (i.e., western North America, South America, South Africa, Australia, New Zealand, China) (Osland et al. 2017b; Cavanaugh et al. 2018; Smith and Sheridan 2020). Compared to their tropical mangrove counterparts within the center of their distribution, mangroves near poleward range limits are morphologically and genetically distinct (Fig. 1). Mangroves near poleward range limits are often shorter, wider, multi-stemmed, and more shrublike

compared to their tropical counterparts that grow as tall forests in freeze-free, resource-rich environments (Perry and Mendelsohn 2009; Dangremond and Feller 2016; Yando et al. 2016; McKee and Vervaeke 2018; Kennedy et al. 2020b, 2022). Thus, there is a need for customized methods and improved temporal and spatial data regarding range limit dynamics in these highly valuable wetland ecosystems (Ximenes et al. 2022).

There are many parallels between mangrove range expansion and woody plant encroachment into terrestrial grasslands, which is a phenomenon that is occurring globally in many different environments (Van Auken 2009; Eldridge et al. 2011; Archer et al. 2017; Huang et al. 2020). Many of the data needs and challenges identified for the mangrove-marsh transition zone are similar to those that have challenged geographers and ecologists working across other forest-grassland ecotones in terrestrial environments (e.g., Gosz 1993; Risser 1995; Browning et al. 2008; Staver et al. 2011). Thus, insights gained regarding patterns, mechanisms, and effects of mangrove expansion are potentially applicable to other forest-grassland ecotones and vice versa (e.g., D’Odorico et al. 2013).

Our overarching objectives in this study were to (1) assess the distribution of mangroves near rapidly changing range limits in the southeastern USA; and (2) identify data limitations and highlight knowledge gaps for advancing our understanding of past, current, and future mangrove range dynamics. We hypothesized that expert elicitation would provide an efficient and cost-effective method for synthesizing this information and providing a foundation from which to expand our knowledge of this dynamic range expansion. With respect to mangrove distribution, we expected that expert elicitation would produce a mangrove presence dataset that was better than the available remote sensing-based products for this region, which tend to underestimate or overestimate mangrove presence in certain areas. To test the efficacy of this practice, we held a virtual workshop in October 2021 that gathered 52 coastal scientists from academic, governmental, and non-governmental organizations across the five states that span the mangrove range limit in the southeastern USA (i.e., TX, LA, MS, AL, and FL). Workshop participants were identified and invited based on their active, on-the-ground mangrove research in this region as evidenced by the published literature, providing the necessary expertise to address our objectives. In the workshop, we used expert elicitation techniques to gather data on mangrove presence and absence throughout the region, as well as to collate information on data limitations and tools needed to advance our understanding of mangrove distribution and range dynamics. We then synthesized these results, following up with additional resource managers, scientists, and extension agents throughout the region as needed to fill in gaps in geographic coverage or content expertise.

Fig. 1 Mangroves growing near freeze-prone range limits are often shorter, wider, multi-stemmed, and more shrublike (see shrubs in upper and middle panels) compared to their tropical counterparts growing in freeze-free, resource-rich environments (see trees in lower panels). Due to these morphological differences, methodological refinements are needed to adequately characterize the structure, function, and abundance of mangroves near poleward range limits. *Avicennia germinans* (black mangrove) is the species shown in the upper and middle panels. The lower panels also include the other two common mangrove species in this region—*Rhizophora mangle* (red mangrove) and *Laguncularia racemosa* (white mangrove). The damage and recovery in the upper left panel are due to a freeze event. Photo credits: Michael Osland



Methods

Focal System: Mangroves in the Southeastern USA

In eastern North America, the northern range limits of mangrove forests are located along the northern Gulf of Mexico and Atlantic coasts of the southeastern USA (Fig. 2). There are three mangrove species in this region that are particularly abundant: the black mangrove (*Avicennia germinans*), the red mangrove (*Rhizophora mangle*), and the white mangrove (*Laguncularia racemosa*) (Odum et al. 1982; Tomlinson 1986). There are other less common plant species often considered mangroves or mangrove associates; for example, the buttonwood mangrove (*Conocarpus erectus*) and the golden leather fern (*Acrostichum aureum*). The

frequency and intensity of extreme cold events govern the northern range limits of all three species, as extreme winter temperatures (i.e., freezing and chilling events) can lead to varying degrees of mangrove damage and mortality (Fig. 1) (Lugo and Patterson-Zucca 1977; Lonard and Judd 1991; Ross et al. 2009; Pickens and Hester 2011; Cook-Patton et al. 2015; Osland et al. 2020a; Bardou et al. 2021). For coastal wetlands located in the tropical-temperate transition zone, winter temperature regimes can produce dynamic forest-grassland ecotones, where woody mangrove plants dominate coastal saline wetlands along warmer coasts (i.e., the tropical side of the biome ecotone, sensu Gosz 1993), and salt marsh plants (i.e., graminoid and succulent plants) dominate coastal wetlands along colder coasts (i.e., the temperate side of the biome ecotone) (Stevens et al. 2006). Mangrove expansion

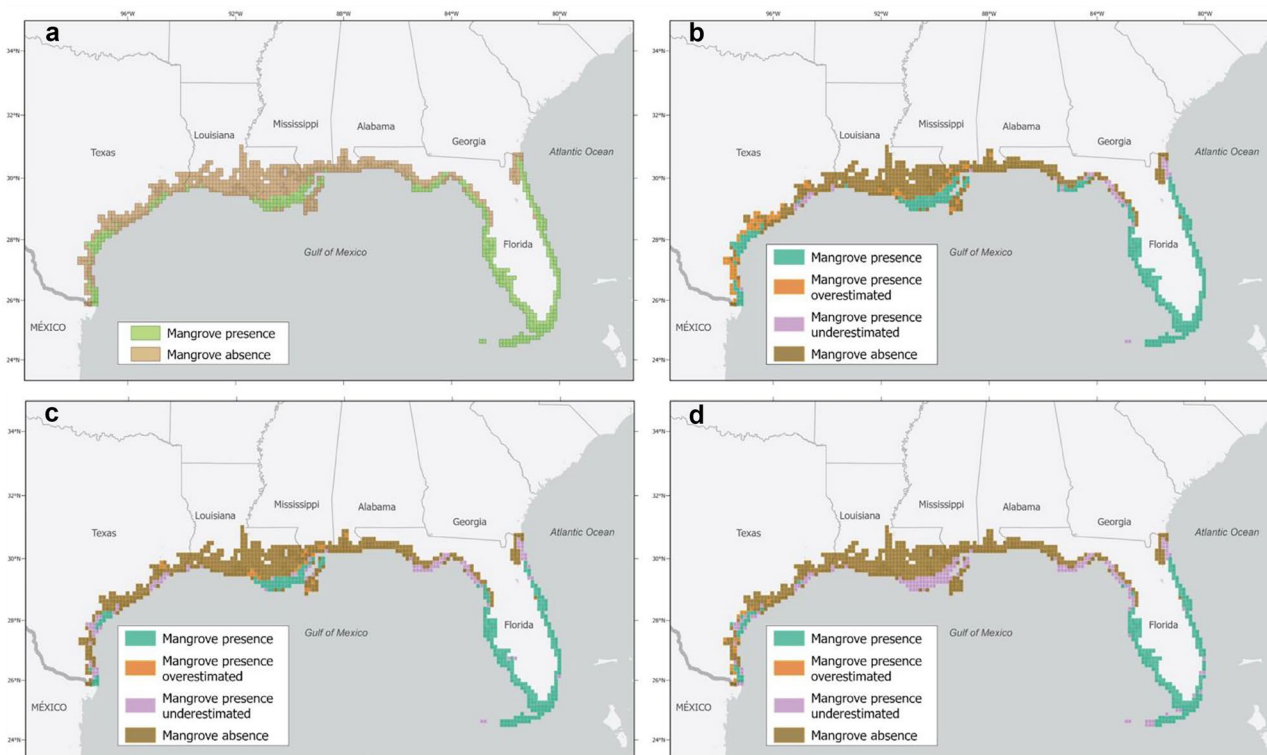


Fig. 2 General mangrove distribution in the southeastern USA. **a** Current distribution of mangroves in the southeastern USA derived from expert knowledge. The other three panels compare the expert-derived distribution with **b** a compilation of all existing distribution

datasets; **c** a compilation of existing global datasets; and **d** state-specific datasets most often used for conservation planning and management purposes

into marsh in this region occurs during freeze-free periods, and, conversely, extreme freeze events can lead to landscape-scale mangrove dieback followed by increases in salt marsh abundance or erosion, where revegetation after mangrove dieback is impaired (Worthy 1990; McBride and Byrnes 1997; Penland et al. 2003).

Since the end of last glacial maximum (~ 19,000 years ago), mangroves have expanded poleward from latitudes closer to the equator due to warming temperatures (Sherrod and McMillan 1985; Woodroffe and Grindrod 1991; Kennedy et al. 2016). Near their current northern range limits, historical analyses of temperature data, aerial imagery, and written historical accounts have revealed many expansion and contraction cycles (Beans 2018). For example, analyses in LA (Osland et al. 2017a) and FL (Stevens et al. 2006; Rodriguez et al. 2016; Cavanaugh et al. 2019; McClenachan et al. 2021) have identified winter temperature-driven cycles of mangrove expansion and contraction during the past century. More recently, mangrove forests have been expanding, both laterally into the marsh and poleward, at the expense of salt marsh within expansion hotspots in FL (Cavanaugh et al. 2014; Snyder et al. 2021b; Vervaeke et al. 2021) and LA (Osland et al. 2017a; McKee and Vervaeke 2018) for about three decades, since the last major region-wide freeze events

caused landscape-scale mangrove mortality in the 1980s. A similar three-decade-long mangrove expansion period in TX (Armitage et al. 2015; Brown et al. 2016) was recently disrupted by an extreme freeze event in February 2021, which caused widespread damage and mangrove mortality. Following such extreme freeze events, salt marsh plants can become more abundant until the next freeze-free period gradually enables mangrove encroachment. Beyond just warming temperatures, interactions with other abiotic drivers (e.g., sea-level rise, hydrologic alterations, hurricanes, fire) can also facilitate mangrove encroachment in this region (Krauss et al. 2011; Raabe et al. 2012; Smith et al. 2013; Van der Stocken et al. 2019a; Kennedy et al. 2020a).

Comparing Distribution Data for Mangroves Near Rapidly Changing Range Limits

When comparing maps and distribution data, it is important to recognize the data's intent and purpose. In our comparisons of mangrove distribution data, we acknowledge and evaluate the differing objectives associated with data creation. Within our study region, reliable mangrove distribution data are more readily available for tall continuous mangrove forest closer to the center of their distribution (e.g.,

south and central FL, Mexico, Cuba), but ground-truthed, reliable mangrove distribution data are not readily available near range limits in north FL, LA, and TX. To illustrate the limitations of existing distribution data for mangroves near the northern range limits in the southeastern USA, we compared an expert-derived generalized mangrove presence dataset (Bardou et al. 2022) to a compilation of existing distribution data. First, we divided the coastal study area into a grid of 0.125° cells (approximately 14 × 16 km) (Fig. 2). For the inland extent, we used the National Oceanic and Atmospheric Administration's [NOAA] Medium Resolution Shoreline dataset (<https://shoreline.noaa.gov/data/datasheets/medres.html>) to include cells that were within 100 m from coastlines and waterways. For each grid cell, we determined mangrove presence or absence using two approaches. First, we collated all available mangrove distribution data in the region, which included global-, state-, and local-scale data and information from the following sources: Sherrod and McMillan (1981; 1985), Zomlefer et al. (2006), Finkbeiner et al. (2009), Spalding et al. (2010), Giri et al. (2011), Scheffel et al. (2013), Elliott (2014), Williams et al. (2014), Armitage et al. (2015), FDEP (2019), Osland et al. (2017a), Bunting et al. (2018), Day et al. (2020), Enwright et al. (2020), McKee et al. (2020), and Snyder et al. (2021b). Global-scale datasets are those that use satellite-derived data to map mangroves at the global scale (Spalding et al. 2010; Giri et al. 2011; Bunting et al. 2018). State-scale datasets are those that characterize mangrove distribution at the state level, either using imagery-based data (Elliott 2014; Armitage et al. 2015; FDEP 2019) or aircraft-based observations (Day et al. 2020). Local-scale datasets are those that characterize mangrove distribution at a sub-state scale using either imagery-based data (Sherrod and McMillan 1981; Finkbeiner et al. 2009; Brown et al. 2016; Osland et al. 2017a; Enwright et al. 2020) or field-based observations (Sherrod and McMillan 1985; Zomlefer et al. 2006; Scheffel et al. 2013; Williams et al. 2014; McKee et al. 2020; Snyder et al. 2021a; Bardou et al. 2022). Next, we classified each grid cell using expert knowledge of known mangrove presence or absence (Fig. 2a). Mangrove presence within these 0.125° cells was defined as having at least one mangrove individually. We compared our mangrove presence grid to the global-, state-, and local-scale data, quantifying the number of cells where the corresponding data indicated mangrove presence (agreement), overestimated mangrove presence (overestimated), or underestimated mangrove presence (underestimated).

Synthesizing Data Limitations and Knowledge Gaps

We asked workshop participants to respond to the following three questions in breakout groups: (1) What are the

limitations of the current data on mangrove distributions? (2) What data are needed to improve our understanding of mangrove distributions and dynamics in the past, present, and future? and (3) What tools are needed to collect these data on mangrove distributions and dynamics? We used coding analyses (Campbell et al. 2013) to summarize workshop participant responses. We developed a coding scheme based on the responses, organizing them first into “basic” themes, then clustering those into “organizing” themes, and further grouping those under “global” themes (Table 1; Armbrorst 2017). We ultimately coded the data at the organizing theme level, which allowed for greater understanding of the data than at the global-theme level while still retaining sufficient intercoder reliability (Campbell et al. 2013). Each of the two coders (i.e., ARH and MJO) analyzed the responses independently and then met to discuss and resolve inconsistencies (Campbell et al. 2013). Once the data were coded, we examined the most frequent organizing themes in response to each workshop question, and we analyzed the thematic proximity of the four to five most frequent codes for each question using the t-coefficient (Armbrorst 2017). This coefficient ranges between 0 and 1 and indicates the average proportion of statements that two themes share with each other (Armbrorst 2017). Our coding scheme had six global themes, 12 organizing themes, and 34 basic themes (Table 1). Intercoder reliability for the organizing themes ranged from 81 to 100% across the three questions (Table 1).

Results and Discussion

Expert Elicitation Generates Updated Mangrove Presence Data that Provides a Foundation for Future Work

The need for improved distribution data for mangroves near range limits is highlighted by our comparison of expert-derived mangrove presence data to existing products (Fig. 2b–d). For example, using a compilation of all available distribution data across the region, we identified 70 cells where mangrove presence was overestimated by existing distribution data and 43 cells where mangrove presence was underestimated by existing distribution data (Fig. 2b, Table 2). While there was full agreement closer to the range core (e.g., south and central FL), there were many areas near the critically important range edges in north FL, LA, and TX where the existing datasets did not accurately depict mangrove presence. In the subsequent paragraphs, we discuss the limitations of global, national, and state-based datasets and, when applicable, general mapping methodology details (e.g., source data, spatial resolution, and mapping approach). Collectively, these comparisons show that there is need for improved spatial data regarding the distribution

Table 1 Coding scheme used to analyze expert responses to three questions: (Q1) What are the limitations of the current data on mangrove distributions; (Q2) What data are needed to improve our understanding of mangrove distributions and dynamics in the past, present, and future; and (Q3) What tools are needed to collect these data on mangrove

distributions and dynamics? Overall reliability refers to the degree of intercoder agreement for the organizing theme in the coding analysis. The Q1, Q2, Q3, and total columns provide the number of times that the organizing theme was included in participant responses for question 1, question 2, question 3, and all combined, respectively

Global themes	Organizing themes	Basic themes	Overall reliability	Q1	Q2	Q3	Total
Global change	Global change interactions	Land use change; sea-level rise; Interactions among global change factors	0.87	5	8	2	15
	Extreme events	Storms; freezes; drought	0.95	11	9	2	22
Sampling constraints	Methodological constraints	Data sharing/storage; new method development; standardized methods; ground truthing	0.98	11	10	22	43
	Logistical constraints	Logistical constraints	1.00	9	0	1	10
	Resource constraints	Financial constraints; personnel constraints	1.00	2	1	3	6
Sampling resolution	Spatial resolution	Spatial resolution	1.00	11	4	5	20
	Temporal resolution	Temporal resolution; historical/baseline data	1.00	18	13	12	43
Ecological mechanisms	Biotic interactions	Species interactions; dispersal	0.81	4	18	4	26
	Abiotic interactions	Suitable habitat; abiotic effects; thresholds; disturbance/resilience	0.85	3	19	5	27
Mangrove traits	Mangrove structure and traits	Species identity; life stage; size/stature/abundance; reproduction; condition	1.00	13	12	3	28
Sampling tools	Remote-based sampling tools	Drones; aircraft; historical photos; remote sensing	1.00	6	6	13	25
	Ground-based sampling tools	Field surveys; pollen/sediment records; citizen science	1.00	4	6	18	28

of mangroves near range limits in the southeastern USA, and that expert elicitation is an effective method for generating general mangrove presence data, which can highlight areas where higher spatial and temporal resolution data are needed. However, these comparisons of existing data are not presented as criticism. All maps and distribution datasets have an intended purpose. For most of these datasets, the intended purpose is not to map mangroves near range limits in the southeastern USA. Thus, we cannot expect these data to be perfect for that purpose. Our intent with this comparison is to highlight the need for improved distribution data,

which are created specifically for mangrove range limits in the southeastern USA.

Global mangrove distribution data have played a critical role in addressing many different questions for mangroves, from identifying areas where mangroves are being converted to other land uses (e.g., Giri et al. 2011; Hamilton and Casey 2016), to estimating global mangrove carbon stocks (e.g., Hamilton and Friess 2018; Rovai et al. 2018), to quantifying the influence of macroclimatic drivers on mangrove distribution (e.g., Osland et al. 2017b; Cavanaugh et al. 2018). However, these data were developed primarily to map mangroves

Table 2 Comparison of expert-derived mangrove distribution data (Fig. 2a) to three kinds of existing distribution data: (1) all available distribution data (left columns; Fig. 2b); global-scale mangrove distribution data (middle columns; Fig. 2c); and state-developed data

(right columns; Fig. 2d). The numbers in columns represent the number of cells where the corresponding data correctly indicated mangrove presence (agreement), overestimated mangrove presence (overestimated), or underestimated mangrove presence (underestimated)

	All data			Global data			State data		
	Agreement	Overestimated	Underestimated	Agreement	Overestimated	Underestimated	Agreement	Overestimated	Underestimated
TX	40	51	12	17	3	35	24	14	28
LA	59	15	0	49	13	10	0	0	59
MS	0	3	1	0	3	1	0	0	1
AL	0	1	0	0	1	0	0	0	0
FL	215	0	30	194	0	51	178	0	67
Total	314	70	43	260	20	97	202	14	155

across the globe and not specifically for patchy mangrove shrubs within a marsh-mangrove mosaic near range limits (Fig. 1) (Ximenes et al. 2022). Global mangrove maps are commonly developed using supervised (i.e., maps developed from training data) and unsupervised classification (i.e., maps developed from automated clustering of pixels) of moderate resolution satellite imagery (i.e., 10–30 m). The coarse spatial resolution of imagery limits the ability for these products to capture sparse, shrubby mangroves that are often found near poleward mangrove range limits (Ximenes et al. 2022). Additionally, these products often falsely identify some plants as mangroves, in particular non-mangrove coastal woody plants and upland plants along higher elevation spoil banks or topographic ridges. Thus, these global data are typically not meant for quantifying the distribution of mangroves near poleward range limits. In our comparison, we identified 20 cells in the southeastern USA where mangrove presence near range limits was overestimated by existing global datasets and 97 cells where mangrove presence near range limits was underestimated by those existing global data (Fig. 2c, Table 2).

In the USA, coastal planners and resource managers typically utilize specific national and state-developed datasets to inform conservation planning and management decisions. Mangroves are most abundant in Florida, which is the only state in this region that has produced a statewide land cover classification system that includes dedicated, reliable mangrove classes [i.e., the “Mangrove Swamp” and “Scrub Mangrove” classes within the Florida Land Use Classification System (Kawula and Redner 2018) and the Florida Land Use, Cover, and Forms Cover Classification System (Florida Department of Transportation 1999; FDEP 2019)]. This land cover product is produced using expert-based photointerpretation and delineation using high-resolution aerial imagery and other ancillary data. This is a valuable land cover classification category that has been used to address a wide range of questions regarding the distribution of mangroves in FL (see discussion in Radabaugh et al. 2017). However, these data are most valuable in south and central FL, closest to the mangrove distributional center. Like other datasets, the Florida Land Cover Classification System was not developed to adequately capture range edge mangroves in north FL (e.g., near Apalachicola or St. Augustine along the Gulf of Mexico and Atlantic coasts, respectively), where there is often a mangrove-marsh mosaic that falls between these two land cover classes (Doughty et al. 2016; Radabaugh et al. 2017; Simpson et al. 2017; Coldren et al. 2019; Snyder et al. 2021b; Vervaeke et al. 2021). The minimum mapping unit for wetlands in FL’s water management district land use/land cover maps that comprise the statewide mapping layer (FDEP 2019) is 0.8 ha (8000 m²). Thus, if patches of mangroves do not exceed this minimum mapping unit,

they are mapped as salt marshes, which underestimates true mangrove extent. We identified 0 cells where mangrove presence near range limits was overestimated by the statewide mapping layer (FDEP 2019) and 67 cells where mangrove presence was underestimated by that data (see FL in Fig. 2d, Table 2).

To the west of FL in the northern Gulf of Mexico, there are large mangrove populations in LA and TX. However, the mainland coasts of AL and MS have historically been too cold to support mangrove populations. To our knowledge, there are currently no mangroves present in Alabama. There are also no mangroves on the mainland MS coast. However, isolated mangrove individuals have been observed on several of MS’s barrier islands (Scheffel et al. 2013; Macy et al. 2019).

In Louisiana, mangroves are most abundant along the southeastern outer coast, which is warmer than interior wetlands or those along the Chenier Plain (Osland et al. 2017a, 2020b; Day et al. 2020). LA does not have a statewide land cover classification system that has a dedicated land cover class for mangroves. However, LA does have an approach for characterizing and monitoring wetland vegetation types within the coastal zone via helicopter landings at points on a grid (Visser et al. 1998, 2000; Sasser et al. 2014), which are then used for generalized map development (Sasser et al. 2014). There are also data that characterize the point-based distribution of mangroves across the entire state via low-level (15–30 m above the ground) aerial surveys by observers in a small fixed-wing airplane (Day et al. 2020). Although mangroves have been present in Louisiana since at least the 1700s, the distribution of mangroves in the state has historically been spatiotemporally dynamic and small in comparison to marshes (Osland et al. 2017a, 2020b; McKee and Vervaeke 2018; Day et al. 2020). Most saline wetlands in LA are dominated by marsh grasses and forbs. Thus, in LA’s coastal planning and management efforts, mangroves have historically been lumped with marsh plants within a “Saline Marsh” category, which is a salt-tolerance-based vegetation class that is defined by dominance by two marsh grass species (*Spartina alterniflora* and *Distichlis spicata*) and/or mangroves (*A. germinans*) (Sasser et al. 2014). We identified 59 cells where mangrove presence is not specifically delineated in the LA wetland classification system due to the inclusion of mangroves within the Saline Marsh class, where it cannot be distinguished from the marsh grasses *S. alterniflora* and *D. spicata* (see Louisiana in Fig. 2d).

Mangrove classification in TX can be challenging because terrestrial shrublands are more common along the southern and central TX coast due to the region’s lower rainfall and arid climate (Everitt et al. 2002; Elliott 2014). In TX, there is a statewide land cover classification system called the Ecological Mapping Systems of Texas (Elliott 2014) that is often used for conservation planning and management purposes. These data are produced using a supervised classification

of land use/land cover from high-resolution aerial imagery, moderate-resolution satellite data, and other ancillary data. These data include a dedicated mangrove class (i.e., the Mangrove Shrubland class). However, this class is not reliable as it incorporates many inland terrestrial shrublands as mangroves. We identified 14 cells where mangrove presence was overestimated by the Texas Ecological Mapping System data and 28 cells where mangrove presence was underestimated in that data (see TX in Fig. 2d, Table 2).

At the national scale, there are several land cover datasets that are used to evaluate wetland change (NOAA 2021; US Fish and Wildlife Service 2021). These are valuable datasets that have been used to address critical questions, ranging from where wetland loss is occurring (e.g., Dahl 1990) to where wetlands need to be protected from land use change (Tiner 2016). However, none of these datasets were developed to characterize mangroves near range limits. For coastal wetlands, the two most frequently used national datasets are (1) the US Fish and Wildlife Service's National Wetlands Inventory (NWI) (U.S. Fish and Wildlife Service 2021); and (2) NOAA's Coastal Change Analysis Program (C-CAP) (NOAA 2021). The National Wetlands Inventory is developed using data and methods such as those used to develop the FL landcover data. In contrast, C-CAP is produced using data and methods similar to those used to develop the global mangrove datasets. Within both the NWI and C-CAP data, there are Estuarine Scrub-Shrub Wetland and Estuarine Forested Wetland classes (sensu Cowardin et al. 1979), which can potentially be used to identify mangroves closer to the center of their distribution (e.g., south and central FL). However, these classes are not reliable indicators of mangrove distribution near range limits in north FL, LA, and TXs. One reason is that the Estuarine Scrub-Shrub Wetland class in the northern Gulf of Mexico can include woody wetland plants that are not mangroves (e.g., *Baccharis halimifolia*, *Iva frutescens*) (Day et al. 2018; Goodin et al. 2018). Additionally, these data are typically not developed with the aim of capturing patchy mangroves near range limits, especially if they are in a marsh-mangrove mosaic. The classification system uses a threshold of 30% coverage for the dominant vegetation type, so polygons that are classified as estuarine emergent marsh can have up to 29% coverage by mangroves.

Expert Elicitation Underscores the Need for High-Resolution Spatiotemporal Data to Better Understand Range Dynamics, Particularly in Response to Extreme Events

Woody plant encroachment into grasslands is a global phenomenon occurring in many kinds of ecosystems in response to various aspects of global change (Van Auken 2009; D'Odorico et al. 2013; Archer et al. 2017). When studying these dynamic forest-grassland transition zones

(i.e., ecotones), geographers and ecologists have been challenged to develop customized approaches for quantifying spatial and temporal changes in woody plant cover relative to grasslands (e.g., Gosz 1993; Risser 1995; Browning et al. 2008; Staver et al. 2011). Many of the same data requirements, challenges, and methods identified by experts for mangrove range limits are similar to those identified for terrestrial forest-grassland ecotones.

Mangrove experts emphasized the need for data that can be used to advance knowledge of temporal changes in mangrove abundance near range limits. The temporal resolution of data was the most frequently noted theme regarding current data limitations (Fig. 3a) and was among the top themes for data needed to improve our understanding of mangrove distributions and dynamics (Fig. 3b). Mangrove abundance near range limits is greatly influenced by the frequency and intensity of extreme freezing and chilling events (Cavanaugh et al. 2019; Osland et al. 2020b; Cohen et al. 2021; Snyder et al. 2021b), which means that changes in mangrove abundance near range limits are typically more abrupt, frequent, and intense as compared to those at the tropical center of their distribution. The need to understand the influence of extreme events (e.g., freeze events, droughts, and storms) that can drive mangrove distribution dynamics in portions of the transition zone (Gabler et al. 2017; Sippo et al. 2018) was reflected as an important theme by the number of statements regarding limitations of current data (Fig. 3a) and the need for improved temporal resolution within the context of extreme events (Table S1). In the face of accelerating climate change, there is an increasing need for ground-truthed, high-resolution spatiotemporal data that can be used to quantify changes in mangrove abundance and structure in the mangrove-marsh transition zone (Ximenes et al. 2022).

Mangroves Near Range Limits Need to Be Studied Differently than Their Tropical Counterparts

Another consistent message from the workshop and our follow-up conversations was that mangroves near poleward range limits are different than their tropical and subtropical counterparts, and that these differences in mangrove structure and traits need to be accounted for in efforts to investigate spatiotemporal dynamics near range limits (Fig. 3a, 3b, and Table S1). Cold-hardy mangroves (primarily *A. germinans*) growing near freeze-prone range limits are often shorter, wider, multi-stemmed, and more shrub-like compared to their tropical tree counterparts growing in freeze-free, resource-rich environments (compare upper and middle panels in Fig. 1 to bottom panels) (Morrisey et al. 2010; Osland et al. 2014). Furthermore, near some range limit areas (e.g., near Port Fourchon, LA), mangrove density can be high and expansive, forming continuous 1- to 2-m-tall mangrove stands that cover large areas (Guo

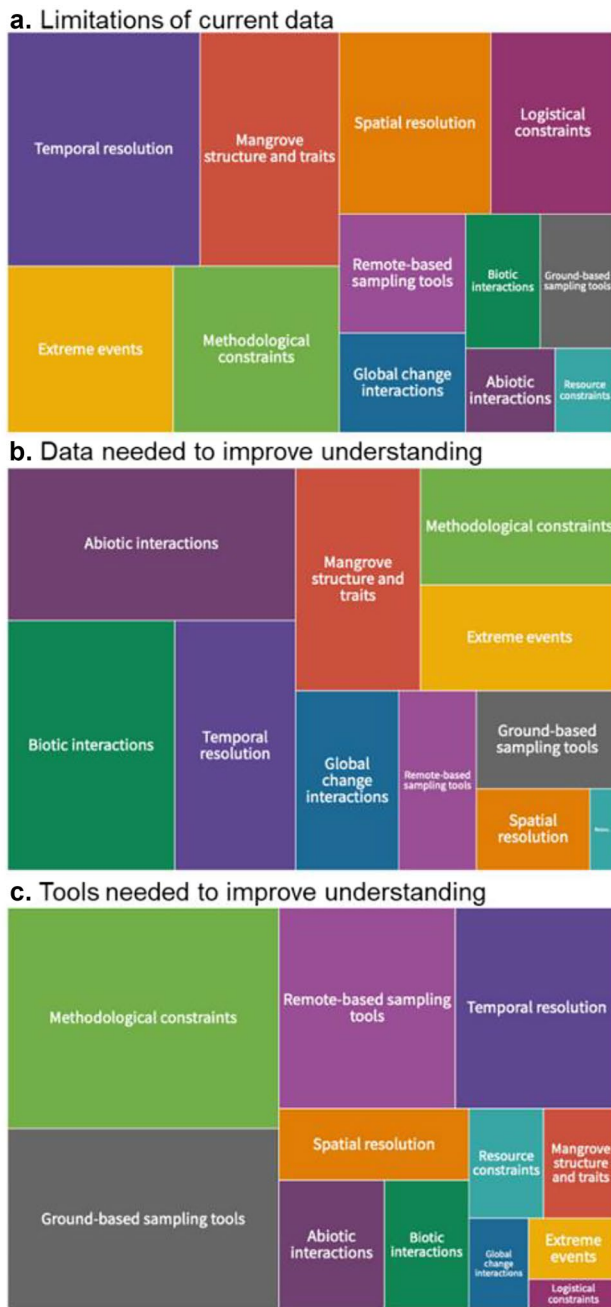


Fig. 3 Recurring themes from expert elicitation. We asked workshop participants to respond to the following three questions: (1) What are the limitations of the current data on mangrove distributions; (2) What data are needed to improve our understanding of mangrove distributions and dynamics in the past, present, and future; and (3) What tools are needed to collect these data on mangrove distributions and dynamics? We organized workshop responses into common themes using coding analysis (Table 1). The size of theme’s rectangle reflects the number of times it was mentioned by participants. Text in the teal box of panel b is “resource constraints”

et al. 2017; Weaver and Armitage 2018; Charles et al. 2020; Macy et al. 2020). However, near other range limit areas (e.g., between Leeville and Golden Meadow, LA), mangrove

densities can be low, producing mangrove-marsh mosaics where mangrove shrub canopy cover may be small, discontinuous, and surrounded by marsh (Day et al. 2020; Snyder et al. 2021b). The shrubby structure and patchy distribution of mangroves in these locations makes them difficult to detect using moderate-resolution (e.g., 10–30 m) datasets and remote-sensing approaches that are often used to produce global mangrove datasets (Giri et al. 2011). Due to these morphological differences, methodological refinements are required to adequately characterize the structure, function, and abundance of mangroves near poleward range limits. For example, the allometric equations needed for calculating the biomass of near-range limit mangrove shrubs (Woodroffe 1985; Osland et al. 2014) are different from the equations needed for tropical trees (Smith III and Whelan 2006; Komiyama et al. 2008). Beyond differences in biomass, the sparse occurrences of individual mangrove plants or seedlings are important to define the leading edge near the range limit. However, these patchy mangroves may fall outside the minimum criteria to be considered mangrove wetland cover from interpretation of remotely sensed data.

Biotic and Abiotic Interactions Can Strongly Influence Range Edge Dynamics

Abiotic and biotic interactions were among the most frequent themes that arose regarding the data needed to improve our understanding of mangrove distributions and dynamics (Fig. 3b), partly due to studies indicating such processes can have large effects on mangrove range expansion. Data on mass dispersal mechanisms are particularly lacking, especially at the range edges where such processes are a key part of mangrove poleward migration (Stevens et al. 2006; Alleman and Hester 2011; Peterson and Bell 2012, 2015; Van der Stocken et al. 2019b; Yando et al. 2021). There is a need to better understand hydrodynamics and other factors related to propagule dispersal, including long-distance mass dispersal events driven by tropical storms (Kennedy et al. 2016, 2020a; Van der Stocken et al. 2019a). Similarly, detailed knowledge of the coastline is needed to determine the availability of habitat for mangrove migration and expansion, as we would not expect mangroves to encroach into all coastal wetlands due to limiting factors such as shoreline energy (Macy et al. 2021), salinity (Visser et al. 1998; Osland et al. 2018), inundation (Krauss et al. 2008; Twilley and Day 2012), geomorphology, edaphic conditions, competition with existing graminoid and succulent plant species (McKee and Rooth 2008; Yando et al. 2018, 2019), or herbivory (Patterson et al. 1997; Langston et al. 2017; Langston and Kaplan 2020; Yando et al. 2021). On-the-ground research could help to better understand competition dynamics and the environment in which mangroves are encroaching into.

Historical Data Can Provide Critical Context for Current Dynamics

Due to the dynamic nature of mangrove range limits in the southeastern USA, there is a need to advance our understanding of mangrove dieback events (range contraction) and mangrove recovery periods (range expansion) that have occurred in the past, as reflected in the importance of methodological constraints as a theme (Fig. 3), which can inform our ability to predict future responses to accelerated warming. Long-term historical perspectives are essential for understanding species response to changes in climate; however, these are often lacking or incomplete (Beans 2018). Deeper understanding of population dynamics can be obtained through non-traditional records such as herbaria, oral histories, historical coastal surveys, indigenous knowledge, and historical literature accounts (Stevens et al. 2006; Osland et al. 2017a, 2020b; Cavanaugh et al. 2019; Snyder et al. 2021b). It can also be obtained through newly acquired data from sedimentary records and paleoecology, which can document long-term dynamics and the timing and history of mangrove populations as well as important records of environmental changes, especially climate and sea-level variations (Yao et al. 2015; Saintilan et al. 2020; Vaughn et al. 2020; Rodrigues et al. 2021; Ryu et al. 2022).

Methodological Constraints Limit Our Current Understanding

Expert consensus was that currently available methods are not sufficient to understand dynamic range-edge populations. The most common theme for tools needed to understand mangrove distributions and dynamics was methodological constraints (Fig. 3c), which is a theme that was closely tied to both ground-based and remote-based sampling tools (Fig. 3c, Table S1). As new remote-sensing techniques are developed, ground truthing is necessary to validate the accuracy of the technique before it becomes widely accepted.

Synthetic aperture radar (SAR) and data fusion (i.e., combining data from multiple remote-sensing platforms) are producing cutting-edge mangrove remote-sensing science. For SAR imagery, L-band data has shown utility in mapping mangroves and characterizing mangrove structure (Simard 2019; Lucas et al. 2020). The availability of L-band SAR data can be a limiting factor, but this will soon change when the National Aeronautics and Space Administration and Indian Space Research Organization launch NISAR (the proposed launch is currently in 2024). This sensor will make L-band SAR freely available globally with a spatial resolution of 3–10 m and a 12-day revisit period. Data fusion of aerial imagery and satellite imagery with elevation data

has been shown to enhance map development and structural estimation (Pham et al. 2019; Lucas et al. 2020).

While the SAR advancements and novel data fusion techniques are exciting and will be useful in areas with continuous mangrove coverage, the spatial resolution of these data may limit their utility for the patchy distribution of mangroves near poleward range limits. Recent improvements in remote-sensing technologies toward very high-spatial resolution and consistent temporal coverage should advance our understanding of range expansion and contraction dynamics. Increasing use of unoccupied aerial systems (UAS) offers a promising mechanism for gathering data from areas that are otherwise difficult to access (Doughty and Cavanaugh 2019; Yin and Wang 2019; Cohen et al. 2021). UAS can be used to collect centimeter-level optical imagery that can be used for mangrove biomass estimation (Doughty et al. 2021) and, potentially, ground truthing of remotely sensed observations such as those from synthetic aperture radar or optical imagery. Structure-from-motion processing can be applied to UAS-based imagery to develop digital surface models that provide information on vegetation height and structure when combined with field data or an existing bare-earth digital elevation model (Enwright et al. 2021). These systems can also be coupled with lidar sensors that can collect very high-resolution point clouds for assessing mangrove structure for individual trees or shrubs (Yin and Wang 2019). This is enabled by data collection with very dense point clouds. For example, depending upon flight planning, point spacing for unmanned aerial lidar systems with current lidar technology can exceed 300 points per m² (Seymour et al. 2018). While traditional airborne lidar can be used to map and quantify biomass for taller mangrove forests (Salum et al. 2020), the point spacing may limit the effectiveness of these data, especially along the poleward range limit where mangroves tend to be sparsely distributed shrubs. Next-generation lidar sensors, such as single photon and Geiger-mode lidar platforms, will enable data collection with higher nominal point spacing (e.g., greater than 20 points per m²) (Stoker et al. 2016), which will increase the utility of non-UAS-based lidar data. However, data from these sensors are not yet widely available. While UAS can provide spatial data with unprecedented detail, the utilization of this technology is currently best suited for site-specific investigations at small spatial scales (e.g., dynamic range expansion hotspots rather than entire states or regions).

Recent advancements in high-resolution commercial satellite imagery enable the production of regional high-resolution mangrove distribution data. These data could be used to develop products at set time intervals along with more frequent maps in more dynamic areas along the poleward range limit. High-resolution satellite imagery combined with high-resolution elevation data could be used to address the data challenges identified through our expert elicitation

related to mangroves being mapped on spoil banks and topographic ridges. For example, researchers have integrated aerial imagery and probabilistic data on tidal regimes when mapping wetland and non-wetland land cover types on barrier islands (Enwright et al. 2019).

Finally, regardless of the data sources, mangrove maps produced along the poleward range limit must attempt to delineate mangrove ecotonal patches rather than generalizing areas as mangrove or marsh that are actually a marsh-mangrove mosaic. Geographic object-based image analysis (GEOBIA; Blaschke et al. 2014) is a powerful and efficient approach that segments high-resolution imagery into objects and then classifies objects instead of individual pixels. GEOBIA has been used for mapping mangroves (Myint et al. 2008) including the identification of individual plant crowns (Kamal et al. 2015). This approach may be used to increase the detail of mangrove maps in areas with sparse mangrove coverage along poleward extents. These various approaches need to be complemented by on-the-ground measurements assessing occurrence in the transition zone, particularly in areas where mangrove density is low or where mangrove seedlings and sapling are obscured by tall marsh plants.

On a broader level, our assessment of existing mangrove maps underscores the impact of methodology on overall map utility. Not surprisingly, the main objectives and budget of a mapping effort play an important factor in the methodology and overall level of detail of the maps. The development of a regional, coordinated, mangrove-specific mapping and field-based vegetation monitoring program for the southeastern USA could be one way to increase the level of detail needed to understand mangrove distribution changes near poleward range limits. Such a program would enable maps and distribution data to be produced for consistent dates with a standardized methodology and could include a hierarchical framework where areas of interest can be mapped at more frequent intervals and in higher spatial resolution.

Crowdsourcing and Regional Data Sharing Have High Potential to Advance Our Understanding

Regional coordinated networks of individuals sharing standardized field-based data offer promise for resolving current logistical constraints and methodological challenges (e.g., Radabaugh et al. 2017; Osland et al. 2020a), particularly when paired with complementary approaches including high-resolution remote-sensing and participatory tools for gathering local knowledge. One prime example of the value of regional coordinated data collection stems from the work of a group called the Mangrove Migration Network, which has utilized coordinated field-based measurements by a network of independent scientists near mangrove range limits in Texas, Louisiana, Mississippi, Florida, and Tamaulipas (Mexico) to refine temperature thresholds for

black mangrove damage, mortality, and recovery (Osland et al. 2020a). These collaborations have answered questions that could never have been addressed without regional collaboration and data sharing. Another example is the Mangrove Survey Network, which is a group of individuals using standardized methods to collaboratively track the expansion of mangroves along the northwestern Gulf of Mexico in MS, AL, and northwest FL (Snyder et al. 2021a; Shepard et al. 2022). Critical questions regarding changes in mangrove abundance and structure near mangrove-marsh ecotones can be investigated with coordinated, standardized field-based vegetation data collections near dynamic range limits in TX, LA, MS, and FL (e.g., Yando et al. 2016, 2018). Field-based approaches by ecologists, conducted in collaboration with remote-sensing specialists, can be used to validate ground-truth products derived from remote sensing.

Workshop participants were particularly excited by the potential for using citizen-science techniques to address challenges of spatial and temporal resolution. Several participants noted the need for a social network application for citizen scientists to photograph and upload georeferenced images of mangroves. Some citizen science-derived mangrove observations are available via iNaturalist (<https://www.inaturalist.org>). However, those efforts could be better coordinated, leveraged, and expanded. Within this region, there is a citizen science project focused on plastic pellet observations (i.e., the “Nurdle Patrol”; Tunnell et al. 2020) that has been particularly effective and could serve as a template for organizing citizen science mangrove observations near range limits.

Conclusions

Our efforts show that expert elicitation provides a feasible and effective approach for advancing our understanding of species’ changing range limits in response to climate change, which are occurring across taxa and systems around the world (Pecl et al. 2017; Henry and Sorte 2021). Although our efforts in this project have focused primarily on mangroves, the insights can be applied to other range-shifting species, since empirical data are often not available to fully capture range dynamics at landscape or regional scales. Such information is particularly critical for foundation plant species such as mangroves (Ellison et al. 2005; Ellison 2019), as it provides a basis for understanding the cascading ecological effects of range expansion on the human communities that depend on their ecosystem services (Kelleway et al. 2017; Armitage et al. 2021; Osland et al. 2022). Furthermore, this understanding is necessary for the design and implementation of effective management plans to address anticipated positive and negative impacts of continued changes in species distributions (Henry and Sorte 2021).

Acknowledgements We thank the many individuals that contributed to this review and, especially, the expert-derived mangrove distribution data. Mark Hester and Ken Dunton provided valuable information for Louisiana and Texas, respectively. This work was supported primarily by a National Academy of Sciences, Engineering, and Medicine Gulf Research Program Healthy Ecosystems Grant to ARH, SBS, CCS, and MJO. Additional support for MJO was provided by the U.S. Geological Survey Climate R&D Program, the USGS Greater Everglades Priority Ecosystem Science Program, and the USGS Southeast Climate Adaptation Science Center, which is managed by the USGS National Climate Adaptation Science Center. Support for AMM was provided by Department of Energy TES grant DE-SC0020116. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Author Contribution RB, MJO, and ARH conceived and initiated this review. SBS, CS, RC, SS, ST, KT, SG, KA, and JAI helped plan and coordinate the workshop. All the authors contributed data and information within and outside of the workshop. RB consolidated and analyzed mangrove distribution data with input from MJO and ARH. ARH developed the coding scheme to synthesize workshop responses. RB, MJO, and ARH prepared figures and tables and wrote the first draft of the manuscript. All the authors contributed to subsequent manuscript drafts and gave final approval for publication.

Funding Open access funding provided by Northeastern University Library.

Data Availability The expert-derived mangrove presence dataset has been published as USGS Data Release (Bardou et al. 2022).

Declarations

Conflict of Interest The authors have no competing interests to declare.

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