Ecological islands: conserving biodiversity hotspots in a changing climate

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For decades, botanists have recognized that rare plants are clustered into ecological “islands”: small and isolated habitat patches produced by landscape features such as sinkholes and bedrock outcrops. Insular ecosystems often provide unusually stressful microhabitats for plant growth (due, for example, to their characteristically thin soils, high temperatures, extreme pH, or limited nutrients) to which rare species are specially adapted. Climate-driven changes to these stressors may undermine the competitive advantage of stress-adapted species, allowing them to be displaced by competitors, or may overwhelm their coping strategies altogether. Special features of insular ecosystems – such as extreme habitat fragmentation and association with unusual landscape features – could also affect their climate sensitivity and adaptive capacity. To help predict and manage climate-change impacts, I present a simple conceptual framework based on a synthesis of over 300 site-level studies. Using this framework, conservation efforts can leverage existing ecological knowledge to anticipate habitat changes and design targeted strategies for conserving rare species.

In a nutshell:
- Insular ecosystems are produced by distinctive landscape features such as rock outcrops, sinkholes, cliffs, and springs
- These ecosystems are small, naturally fragmented, and geographically “anchored” within the landscape
- Many insular ecosystems are biodiversity hotspots for rare plants but the impacts of climate change on these systems are still poorly understood
- Habitat suitability for rare, specially adapted plants is maintained by physical stressors that inhibit competitors from the surrounding landscape
- Climate-driven shifts to physical stress regimes are therefore a promising framework in which to anticipate climate-change effects on biodiversity in insular ecosystems, including identification of potential climate microrefugia

From an ecological perspective, an “island” is a “patch of suitable habitat surrounded by unfavorable environment that limits the dispersal of individuals” (Brown 1978). In this review, I define insular ecosystems (Figure 2) as having (1) individual patches that are spatially isolated from one another (“islands”) and embedded within a matrix of a contrasting ecosystem (the “sea”; (2) total spatial areas that are very small (typically <5%) compared to the areas of their surrounding regions; and (3) boundaries marked by steep environmental gradients.

Although many insular ecosystems have received attention for their rare species and unusual habitats, climate-change effects on biodiversity in these ecosystems are still poorly understood (Cartwright and Wolfe 2016). The literature on insular ecosystems is fragmented, with many small site-level studies (see list of references in WebPanel 1) but few synthetic reviews that are limited to a single insular ecosystem (eg limestone glades) or a set of related types (eg various kinds of depression wetlands). This review provides a holistic qualitative synthesis of insular ecosystems to derive common themes across diverse ecosystems and to explore the special features of insular ecosystems that require consideration in climate-change vulnerability assessments. A simple conceptual model is presented that can help to anticipate climate-change effects on plant communities in insular ecosystems. This approach also incorporates non-climate threats to assess overall vulnerability (exposure, sensitivity, and adaptive capacity; Stein et al. 2013). Although insular ecosystems are a global phenomenon, this review uses the southeastern US as a case study, a region rich in rare
plant species that are endemic to insular ecosystems (i.e., confined to a particular ecosystem and requiring that ecosystem for habitat) (Estill and Cruzan 2001; Loehle 2006; Noss 2013). Concepts presented here are based on a synthesis of more than 300 localized botanical and ecological studies conducted in a variety of ecosystems over several decades (WebPanel 1).

### Insular ecosystems as biodiversity hotspots

Biodiversity hotspots – localized concentrations of rare and endemic species – are promising conservation targets to mitigate biodiversity losses resulting from climate change (Myers et al. 2000; Noss et al. 2015). Some insular ecosystems arguably qualify as biodiversity hotspots, as evidenced by richness of endemic and globally rare plant species (Figure 3; WebTable 1; references marked with “B” in WebPanel 1) in ecosystems that encompass very small geographic areas (Collins et al. 2001; Cartwright and Wolfe 2016). Although quantitative analyses of rare plant richness relative to island size are often difficult to calculate (WebPanel 2), botanical accounts commonly emphasize the extraordinary diversity of rare plants in certain insular ecosystems (WebPanel 1). In the eastern US, for example, 11 species are endemic or nearly endemic to sandstone rockhouses (recesses under cliff overhangs; Walck et al. 1996). The Ketona dolomite glades of Bibb County, Alabama, have been described as a “botanical lost world” with at least 60 plant species of conservation concern and nine species found nowhere else on Earth (Allison and Stevens 2001). Although the focus of this review is on plant communities, several insular ecosystems also provide habitat to animals of conservation concern, especially rare invertebrates (Braunschweig et al. 1999; Wiser and White 1999; Cartwright and Wolfe 2016).

Many insular ecosystems also contain disjunct populations of plants (that is, isolated populations that are geographically distant from the primary range of the species), and therefore contribute to regional biodiversity by supporting species that are globally common but regionally rare (Braunschweig et al. 1999; Hill 1999; Wolfe et al. 2004). For instance, the fern Wright’s cliffbrake (*Pellaea wrightiana*) grows primarily in the US Desert Southwest but has disjunct populations more than 1000 km to the east in granite outcrops of North Carolina (Wyatt and Fowler 1977). Similarly, the population of mountain alder (*Alnus viridis crispa*) in grassy balds on Roan Mountain in Tennessee and North Carolina is more than 1000 km from its primary range in boreal and Arctic regions to the north. Some insular ecosystems support globally rare associations (groupings) of plants. For example, an association of overcup oak (*Quercus lyrata*), river birch (*Betula nigra*), and resurrection fern (*Pleopeltis polypodioides*) is known only from a single sinkhole wetland in Tennessee (Wolfe et al. 2004).

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**Figure 1.** Examples of insular ecosystems: (a) granite outcrop, (b) high-elevation grassy bald, (c) sinkhole wetland, and (d) Carolina bay wetland.
Other unique plant associations are restricted to small clusters of sinkhole wetlands or to flood-scoured riparian outcrops (Fleming et al. 2012).

Insular ecosystems are not only biodiversity hotspots but also key contributors to geodiversity (the diversity of geology, soil, and topography across a landscape). Many insular ecosystems are associated with distinctive geologic and topographic features, such as sinkholes, river gorges, rock outcrops, and springs (Collins et al. 2001; Kelso et al. 2001). Insular ecosystems exemplify the critical link between geodiversity and biodiversity, a concept long recognized by botanists (Kruckeberg 1986) and increasingly emphasized in conservation planning (Anderson and Ferree 2010; Comer et al. 2015). Because unusual landscape features may produce regionally rare microenvironments, their inclusion in conservation networks may be particularly important for conserving diversity not only of species but also of ecological and evolutionary processes (Anderson and Ferree 2010; Comer et al. 2015).

**Special features of insular ecosystems**

Although a diverse set of upland and wetland habitats qualify as insular ecosystems, they all share special features that set them apart from more geographically widespread ecosystems, with important implications for climate-change vulnerability and conservation strategies.

**Naturally fragmented habitats**

By their very nature, insular ecosystems represent highly fragmented habitats (ie many small and isolated habitat patches as opposed to large, geographically continuous habitats). Because climate change is driving species’ migrations (Loarie et al. 2009; Corlett and Westcott 2013), relative habitat connectivity along climate gradients is a component of extinction risk (Klausmeyer et al. 2011). Habitat fragmentation is generally regarded as a conservation problem and a by-product of human activities (Haddad et al. 2015). Insular ecosystems complicate these assumptions because their fragmentation is both natural (habitat islands were small and isolated prior to human interference) and human-caused (many islands have been degraded or destroyed; Noss et al. 1995). It is unclear whether maintaining “large, intact landscapes” (Watson et al. 2011) is useful for conserving biodiversity in insular ecosystems. On the one hand, the vast majority of those landscapes do not represent suitable habitat for rare, endemic species; on the other hand, conservation of the surrounding “sea” might deter future destruction of habitat islands and reduce problems associated with anthropogenic fragmentation (eg invasive species’ introductions). Naturally fragmented ecosystems may not be adequately described by the current science of habitat connectivity, and may require specialized conservation strategies (Cartwright and Wolfe 2016) that account for the role of genetic isolation in the evolution and maintenance of biodiversity in these ecosystems (Kruckeberg 1986; Collins et al. 2001).

**Geologic and topographic anchoring**

Predictions of biodiversity loss have emphasized species’ abilities to “keep pace” with changing climate (Loarie et al. 2009; Corlett and Westcott 2013), but approaches assuming steady migration across homogenous landscapes are clearly inappropriate for insular ecosystems (Figure 4; Ibáñez et al. 2006). Indeed, species confined to montane ecosystems have...
been identified as having “nowhere to go” (Wiser and White 1999; Loarie et al. 2009) because they occupy microhabitats at extreme ends of local environmental gradients that are “anchored” in place by geologic and topographic features that cannot move across the landscape. A similar situation confronts ecosystems that have received less attention, such as cliffs, rockhouses, springs, sinkhole wetlands, certain bogs and fens, and barrens defined by particular geologic substrates (eg serpentine and shale). For rare plant communities restricted to such landscape features, migration may require “island hopping”, or long-distance dispersal between isolated and geographically static habitat patches. As such, dispersal modes of species endemic to insular ecosystems may be an important control on adaptive capacity in response to climate change (Beever et al. 2016).

Characteristic stress regimes

Insular ecosystems are generally characterized by high levels of physical stress (including temperature and other factors listed below) relative to their surrounding landscapes (references marked with “S” in WebPanel 1). Stress regimes – characteristic combinations of natural physical (as opposed to biological) stress factors, such as temperature, pH, water availability, soil depth, and disturbance – appear to be vitally important in maintaining habitat for rare and endemic plants by impeding competition. Five general stress-regime categories characterize the environmental conditions across a variety of insular ecosystems (WebTable 2). For example, although granite outcrops, sandstone outcrops, and limestone glades within the southeastern US have different geographies and geologic substrates, they share a common set of stressful conditions: extremely thin soil, scarce shade, and seasonally hot and dry soil conditions. This stress regime has been characterized by direct measurement (eg soil temperatures as high as 50°C and soil moisture below the permanent wilting point; Baskin and Baskin 1999; Braunischweig et al. 1999; Shure 1999). This stress regime is also reflected in specialized adaptations of endemic plants (WebTable 2), several of which resemble desert plants in their physiology even though the southeastern US generally receives abundant rainfall (Shure 1999). Intriguingly, several glade and outcrop ecosystems also include seasonally wet microenvironments in depressions or seepage areas, which support aquatic endemic plants with their own specialized adaptations (WebTable 2; Shure 1999).

Numerous studies of plants endemic to insular ecosystems have demonstrated two common themes (WebPanel 1). First, endemic plants typically have specialized adaptations to tolerate environmentally stressful conditions specific to the ecosystems they occupy. Second, endemic plants are generally poor competitors (ie slow-growing and shade intolerant; Baskin and Baskin 1988).

Endemic plants therefore find competitive advantage in insular ecosystems, where physically stressful conditions inhibit their competitors from the surrounding landscape (Braunschweig et al. 1999; Shure 1999; Cofer et al. 2008). For instance, plants endemic to isolated flood-scarred riparian outcrops have a range of coping strategies, including special anchoring structures, “mechanical fuses” to protect roots, and the ability to use floods to facilitate reproduction (Bailey and Coe 2001). Competitors from the surrounding forest lack these adaptations and are unable to withstand the destructive floods, which maintain the open, sunny habitat islands required by river scour endemics (Vanderhorst et al. 2007; Wolfe et al. 2007).

Climate change in insular ecosystems: gaps in understanding

A coherent picture has emerged regarding the biodiversity, biogeography, and characteristic stress regimes of insular ecosystems based on hundreds of botanical and ecological studies. However, few studies have explicitly addressed climate change (WebPanel 1; Damschen et al. 2012). For models of future climate to be useful in predicting endemic species’ distributions or extinctions in insular ecosystems, they must be appropriate in scale and mechanistically linked to the stress regimes that maintain habitat for endemics. The importance of microscale (<<1-km²) differences in climate exposure based on topography is increasingly recognized in conservation biology (Hannah...
et al. 2014) and may be critical to certain insular ecosystems (e.g., cliffs, rockhouses, high-elevation outcrops).

Insular ecosystems can pose challenges of scale and resolution for modeling approaches because while the overall distribution of islands may encompass hundreds of thousands of square kilometers, individual islands harboring rare plant populations may be smaller than an individual 25-m × 25-m grid cell (Shure 1999; Cartwright and Wolfe 2016). Downscaled climate models rarely reach resolutions finer than 25 m (Potter et al. 2013), and as such they may obscure ecologically important gradients even within larger islands (i.e., up to a few hectares), necessitating site-specific microclimatic models. For example, approaches combining light detection and ranging (lidar)-derived topography with dense networks of climate loggers (George et al. 2015) can achieve sub-meter resolution (Lenoir et al. 2017). Although recent advances in microclimatic modeling have focused primarily on temperature and humidity (Lenoir et al. 2017), other important microclimatic variables include soil moisture, wind speed, snow depth, frost exposure, fog formation, and solar radiation (Potter et al. 2013; Hannah et al. 2014). Microscale differences in substrate geology, geochemistry, soil characteristics, and hydrologic and fire regimes may also play fundamental roles in translating shifts in regional climate into changing microenvironmental conditions shaping species’ distributions. After all, favorable climate (even microclimate) alone cannot predict future habitat locations for species restricted to spring-fed wetlands if the new locations lack springs. Moreover, springs must continue to provide the proper physical stress regimes, requiring analysis of climate controls on spring hydrology and geochemistry. Therefore, scientists must ask not only “how fast can species move?” but also “how will regional climate change affect the island microhabitats and stress regimes that species experience as they move?”

Figure 4. Assumptions of steady migration along climate gradients are likely to be unsuitable for species requiring ecosystems that are geographically anchored to the landscape by particular geologic and topographic features, including (a) cliffs, bluffs, and rockhouses; (b) rock outcrops; (c) springs and seeps; and (d) sinkhole wetlands.

### Anticipating climate-change effects in insular ecosystems

To manage anticipated changes in insular ecosystems, we should first examine how climate change will alter ecologically important stress factors, and how changing stress regimes will affect competitive dynamics of plant communities.

### A conceptual model of stress-regime alteration

A fundamental ecological concept is that “along key environmental gradients, species appear to find one direction to move along” (Sutherland et al. 2014). Assumptions of steady migration along climate gradients are likely to be unsuitable for species requiring ecosystems that are geographically anchored to the landscape by particular geologic and topographic features, including (a) cliffs, bluffs, and rockhouses; (b) rock outcrops; (c) springs and seeps; and (d) sinkhole wetlands.
be physically stressful and the other to be biologically stressful" (Guisan and Thuiller 2005). The horizontal axes in Figure 5 depict this trade-off between biological stress (i.e., competition, on the left) and physical stress (on the right). Grime’s (1977) conceptualization of plant strategies (“Competitive”, “Ruderal”, and “Stress-tolerant”) can help generate and test hypotheses about ecological implications of changing stress regimes. Endemic plants in insular ecosystems are typically “S” strategists: that is, poor competitors that possess specialized adaptations to cope with physical stress. Insular ecosystems typically contain regionally rare microenvironments of high stress and low competition, with habitat suitability for endemic plants maintained by physical stress levels between certain upper and lower bounds (shaded area in Figure 5a). This baseline stress regime results from interactions between regional climate and local microhabitat (e.g., soil thickness, degree of shading, local hydrology and geochemistry).

What if climate change reduces physical stress in an insular ecosystem? For example, warmer growing seasons with more frequent droughts could lower water tables and accelerate decomposition in mountain bogs, possibly reducing such stress factors as acidity, anoxia, and nutrient limitation (Schultheis et al. 2010). This would constitute a leftward shift (Figure 5b), reducing physical stress and increasing competition. Rare, stress-adapted bog plants could be displaced by woody encroachment from stress-intolerant competitors (“C” strategists; Grime 1977) from the surrounding landscape.

Conversely, what if climate change intensifies physical stress? For instance, the same regional warming and drought intensification could make seasonally hot and dry conditions in rock outcrops even more extreme. Greater physical stress would shift the stress regime to the right (Figure 5c), which could overwhelm the coping strategies of even specially adapted endemic species. Notably, this conceptual model suggests that a large enough shift in either direction (i.e., rapidly increased or decreased physical stress) could potentially degrade habitat for rare endemic plants.

**Toward a holistic assessment of ecosystem vulnerability**

This conceptual model of stress-regime alteration (Figure 5) can be integrated into a holistic assessment of ecosystem vulnerability (WebFigure 1), incorporating non-climate threats and the biological and physical characteristics of each ecosystem (Klausmeyer et al. 2011; Watson et al. 2011; Pearson et al. 2014). Management efforts can be informed by anticipating stress-regime alteration and gleaning data from diverse sources (WebTable 3), including site-specific studies from the past several decades (examples in WebPanel 1).

Regional climate change is mediated by microhabitat characteristics to produce localized changes in the physical environment that organisms experience as climate-change exposure (Storlie et al. 2014). These microhabitat changes produce shifts along the spectrum of competition and physical stress (Figure 5), which are mediated by stress-tolerance physiology (relative abilities of endemic plants and their competitors to cope with changing physical stressors) to influence climate-change sensitivity in insular ecosystems. Baseline stress regimes in some insular ecosystems are well characterized (references marked with “S” in WebPanel 1) and sensitivity to stress-regime alteration can be inferred from a variety of sources. For example, thermal tolerance of endemic plant seeds (Platt 1951), demographic simulations (Bernardo et al. 2016), and observed community shifts from human alterations and droughts (Pechmann et al. 1991) all provide clues about potential sensitivity to stress-regime change. In addition, dozens of studies have described non-climate threats (references marked with “T” in WebPanel 1) that may alter species’ sensitivity to microclimatic change or limit the range of adaptive responses (Bellard et al. 2012; Damschen et al. 2012; Beever et al. 2016). Mitigating these non-climate threats may be important for enhancing adaptive capacity and maintaining suitable microhabitats for rare plants.

Application of the framework depicted in WebTable 3 and WebFigure 1 can be illustrated with a hypothetical set of insular wetlands. Existing knowledge of the baseline stress regime
combined with downscaled climate models indicating warmer,
drier growing seasons could suggest shorter flooding duration
and localized changes in soil and water chemistry. Sensitivity
to this changing stress regime, and the resulting changes in
competitive dynamics, would be mediated by stress tolerance
abilities and thresholds of inundation-adapted plants relative
to their upland competitors (Sharitz 2003), and might also be
influenced by non-climate threats such as water pollution or
invasive species.

Some wetland specialists may adapt in place through physi-
ological changes (eg improved efficiency of stress adaptations
or changing seasonal timing of life cycles). These capacities
will be mediated by species’ traits, as well as by population
genetics and demographics. Some species might adapt in place
by shifting toward deeper and more frequently flooded zones,
depending on the availability and diversity of newly suitable
microhabitats. Adaptive capacity via long-distance dispersal to
distant wetlands with more suitable hydrology may be con-
strained by species’ traits (eg dispersal mode) or interspecies
interactions (eg availability of pollinators or presence of patho-
gens). Non-climate threats can also constrain adaptive capac-
ity, such as declining regional wetland connectivity as wetlands
are degraded by peat harvesting, logging, and fire suppression
(Buhlmann et al. 1999; Roberts et al. 2004). Such considera-
tions can help tailor adaptive management plans to the rare
plant populations of each insular ecosystem.

Will insular ecosystems provide climate-change refugia?

Population genetics and paleoecological evidence suggest that
certain insular ecosystems – such as karst depression wet-
lands and high-elevation rock outcrops – sheltered relict
plant populations during previous climatic shifts (Braun
1955; Wiser 1994). Concentrations of endemic species and
relict lineages have been noted across a variety of insular
ecosystems (WebTable 1) and are hallmark characteristics
of paleoclimate refugia (Harrison and Noss 2017). However,
refugia from current anthropogenic climate change may not
be spatially or ecologically equivalent to past climate refugia
(Keppel et al. 2015). Furthermore, a detailed comparison
of the defining characteristics of insular ecosystems and
climate refugia reveals important conceptual differences
(WebTable 4).

The functioning of climate-change refugia depends on
species-specific habitat requirements and life-history traits
(Keppel et al. 2015; McLaughlin et al. 2017; Stralberg et al.
2018). Figure 5 provides a framework to anticipate which
types of insular systems – and perhaps even which individ-
ual islands – might provide climate refugia for which groups
of species. Microclimate stability is a defining feature of cli-
mate refugia (Ashcroft et al. 2012; Morelli et al. 2016), spe-
cifically “stable” refugia (McLaughlin et al. 2017). Hence, an
island will only function as a stable refugium for stress-
adapted endemics – including many rare and threatened
plant species – if the island’s stress regime is relatively steady
through time (minimal horizontal shift in Figure 5a).

However, if physical stress is reduced due to climate change
(Figure 5b), the island may function as a “relative” refugium
(McLaughlin et al. 2017). Specifically, the island may serve
as a refugium for competitors from the surrounding land-
scape (eg upland woody species that encroach into bogs as
water levels decline and soil geochemistry becomes more
similar to that of uplands) but not for stress-adapted endem-
ics (eg displaced rare bog plants that may be vulnerable to
local extirpation). Conversely, sharp increases in physical
stress (Figure 5c) may prevent islands from providing refu-
gia for any vascular plant species. For example, rock out-
crops that are already warmer and drier than their surround-
ings are unlikely to provide refugia from a warming, drying
regional climate, and may in fact become increasingly inhos-
pitable to all vascular plants. Importantly, individual islands
within an insular ecosystem type may vary widely in their
degree of stress-regime change (and therefore their refugial
capacity), given similar exposure to regional climate change.
For instance, some springs may provide stable microrefugia
in a drying climate whereas others nearby may cease flowing
entirely (Cartwright and Johnson 2018).

Conclusions and management implications

Although insular ecosystems are diverse in their natural
histories, geomorphologies, and the species they support,
they share common ecological features that require special
consideration in conservation planning. First, insular eco-
systems commonly represent concentrations of large num-
bers of rare species in small areas (ie biodiversity hotspots;
Myers et al. 2000; Allison and Stevens 2001; Kelso et al.
2001). Second, species endemic to insular ecosystems typ-
ically share risk factors for extinction, including small
and isolated populations, rare microclimates, and highly
specialized habitat requirements (Estill and Cruzan 2001;
Bellard et al. 2012; Pearson et al. 2014). Third, because
the landscape features that create insular ecosystems (eg
cliffs, sinkholes, rock outcrops) contribute to regional
godiversity and microclimate diversity, conservation of
these features may be part of a coarse-filter approach to
biodiversity conservation that is relatively robust to climate
change (Anderson and Ferree 2010; Comer et al. 2015;
Lawler et al. 2015).

The special characteristics of insular ecosystems also sug-
gest that they may require management approaches that dif-
fer from those for large landscapes. For example, managers
may need to consider geographic patterns and spatial
arrangement of islands within an archipelago to promote
microhabitat diversity. Although small islands might seem
expendable, their conservation may still be important, as
they may provide distinct microenvironments and serve as
“stepping stones” to facilitate movement between larger
islands (Sharitz 2003; Cofer et al. 2008; Hannah et al.
2014). In some cases, management decisions may be informed by
the potential for certain insular ecosystems to provide climate-change microrefugia (WebTable 4; Morelli et al. 2016), bearing in mind that refugial capacity may vary among islands within an ecosystem type.

Management options in the face of climate change (e.g., land acquisitions, habitat restoration, species’ reintroductions, assisted colonization of new sites) commonly require managers to predict which landscape locations will provide suitable habitat for species in the future. However, even when evidence-based guidance exists on microhabitat requirements for rare plant conservation and reintroduction, it may not incorporate climate change (e.g., Thompson et al. 2006). To be useful in predicting ecological responses in insular ecosystems, downscaled climate forecasts or site-level microclimate models must be linked to the proximal drivers of ecosystem structure and function, such as characteristic stress regimes that maintain habitat for rare species. Thanks to decades of research on individual sites and species (WebPanel 1), physical stress regimes and competitive dynamics in many insular ecosystems are now well understood (Cartwright and Wolfe 2016). Synthesis across studies reveals that a few general categories of stress regimes are common to many insular ecosystems (WebTable 2).

Using a simple conceptual model of stress-regime alteration (Figure 5) within a holistic framework for ecosystem vulnerability assessment, this existing knowledge can be leveraged into ecological predictions to inform management strategies (WebFigure 1; WebTable 3). For example, if regional climate forecasts and microclimate characteristics indicate that physical stress regimes will weaken (Figure 5b), then managers may employ strategies to maintain habitat for rare plants by targeting their competitors (e.g., removal of woody shrubs that encroach into bogs; Schultheis et al. 2010). Alternatively, if stress regimes are expected to intensify (Figure 5c), then managers might assess the capacity of vulnerable species to move to other microhabitats—other islands—with less extreme stress levels. In either case, long-term monitoring of rare plant populations will continue to be important (Collins et al. 2001) in order to validate climate-change hypotheses and assess the effectiveness of conservation strategies. Similarly, continuing the long legacy of botanical and ecological investigations in these fascinating ecosystems (WebPanel 1) will become increasingly important in the context of accelerating global change.

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Mermaids move inland

Cup-shaped Acetabularia spp, a genus of green algae commonly known as mermaid’s wineglasses, are among the world’s largest single-celled organisms, with “caps” shown here averaging roughly one centimeter in diameter. The genus was famously used to demonstrate the central role of the nucleus in eukaryotic cell biology (Int Rev Cytol 1953; 2: 475–98; doi.org/10.1016/S0074-7696(08)61042-6) but much less is known about the ecology and evolution of Acetabularia and related species like Batophora oerstedii (dark-green “fingers” in the photo on the left). Acetabularia and Batophora are typically associated with tropical or warm-temperate shallow marine habitats such as reefs and estuaries. In the Bahamas, they can also be found in inland ponds, often forming “mats” of vegetation that are much larger and denser than those typical for marine populations. Some mats grow in ponds that experience extreme abiotic fluctuations that vastly exceed marine conditions. For example, salinity at Reckley Hill Settlement Pond on San Salvador Island (where both photos were taken) can vary from one-half to twice the average level for seawater, depending on seasonal precipitation. Yet Acetabularia and Batophora thrive there, forming a nearly unbroken 1.5-km ring of vegetation around the pond’s shoreline that can be several meters wide. How do these algae cope with such extreme environmental change? Do they die back and recolonize from nearby stable habitat refuges, or have they adapted to persist in place? Do the survival strategies of these algae, as primary producers, affect other trophic levels? If so, then inland Bahamian ponds could be natural labs for investigating climate change and other threats to marine macroalgae and their associated communities.

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