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POPULATION VIABILITY ANALYSIS OF THE DWARF SEAHORSE, *HIPPOCAMPUS ZOSTERAE*, IN FLORIDA

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Introduction

The dwarf seahorse, *Hippocampus zosterae*, is a relatively small, short-lived species reaching maturity in about 3 months, exhibiting a protracted reproductive cycle, and living to about 2 years of age (Strawn, 1958; Lourie et al., 2004). In the United States, the dwarf seahorse is believed to occur in seagrass habitats from Texas to the east coast of Florida. Records of dwarf seahorse distribution and abundance suggest the dwarf seahorse is more abundant in bay systems with warmer water temperatures and available seagrass habitats (Lourie et al., 2004; Masonjones et al., 2010; K.L. Smith, unpublished data). Thus, healthy seagrass beds are important given this species' unique life history compounded with its sole reliance on vulnerable nearshore seagrass habitats. The species is currently listed on the International Union for the Conservation of Nature (IUCN) Red List as a species of Least Concern (Masonjones et al. 2017).

Although dwarf seahorse range is believed to include the southeast United States and Gulf of Mexico, the direct harvest is limited to Florida by the Marine Life Industry. In general, harvest by the Marine Life Industry pertains to the non-lethal (i.e., animals must be landed alive) harvest of saltwater fish, invertebrates, and plants for commercial purposes, primarily as ornamentals for the aquarium market. In 2016, the Florida Legislature adopted the requirement of an open-access Marine Life endorsement to the Saltwater Products License and implemented a suite of state-mandated gear types and trip-limits in order to better understand this fishery[1]. In addition to the direct harvest by the Marine Life Industry, there is some concern over the potential for bycatch from the bait-shrimp trawl fishery. There is also concern over the increase in trawling effort during the peak reproductive season of dwarf seahorse given the number removed from the population by non-selective fishing is not monitored (Stallings et al., 2014).

Significant issues have been raised about the species status given its unique life history compounded with its sole reliance on vulnerable nearshore seagrass habitats (United States Government Federal Register 77 FR 26478). One of the primary causes of seagrass loss within the range of dwarf seahorse is harmful algal blooms (HABs). HABs are a source of ecological disturbance that alter water quality and patterns of primary productivity through hypoxia and

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[1] Florida Fish and Wildlife Conservation Commission. 2016. A history of Florida's management of the dwarf seahorse (*Hippocampus zosterae*). Unpublished report presented to National Marine Fisheries Service, St. Petersburg, Florida.

shading. Additionally, acute mortality of fishes can occur as part of a HAB due to the increased presence of *Karenia brevis*, the species that causes the most prevalent HABs off Florida. Some HABs are naturally occurring, but are exacerbated by nutrient runoff from anthropogenic sources. Over the last several decades, the frequency and geographic distribution of HABs have been increasing worldwide (Gilbert et al., 2005; Anderson et al., 2008; Paerl et al., 2018), particularly in the Gulf of Mexico (Heisler et al., 2008).

Assessments of population status are uncommon for seahorses due to the demands of other more important commercial and recreational species such as groupers and snappers. As a result, the dwarf seahorse has not undergone a formal population assessment to determine its status, and it is unlikely any such formal assessment will be conducted. Thus, we conducted a population viability analysis (PVA) to evaluate the status of dwarf seahorse in Florida, by examining the effect of commercial harvest, habitat loss, and potential catastrophes (i.e., harmful algal blooms and acute cold mortality events). We also examined the future status of the dwarf seahorse in Florida exploring both optimistic and pessimistic simulations.

Methods

A deterministic age structured life table and Leslie matrices were developed for 3 age classes of male dwarf seahorse: Age-0 (from birth to just before 1 year old), Age-1, and Age-2. Traditionally, females are modeled in age structured life tables; however, males were modeled because males carry the fertilized eggs in a brood pouch, nourish them throughout gestation, and give birth.

Inputs to the model were derived from data available in the literature and through unpublished reports and personal communications (Table 1). We used adult survival rates from mark-recapture experiments and a Cormack-Jolly-Seber model to calculate an annual survival rate for male Age-1 and Age-2 dwarf seahorse. Estimates of survival (φ) ranged 0.97 to 0.98 (Masonjones et al., 2010). By multiplying estimates of survival by each sampling interval, annual adult survival rate was 0.383 yr^{-1}. As there are no estimates of natural mortality (M) for Age-0 dwarf seahorse, the instantaneous rate of natural mortality (converted to survivorship) was estimated as the average of multiple indirect life-history methods to incorporate the range of mortality mechanisms (e.g. temperature-dependent, age scheduling, and weight). Hoenig's (1983) method relies on estimates of longevity; whereas Pauly's (1980) method uses parameters estimated through the von Bertalanffy growth model (calculated in this study using size and growth information from Strawn (1958) and Azzarello (1990) and water temperature). The methods of Peterson and Wroblewski (1984) and Lorenzen (1996) estimate M based on body mass-at-age and Jensen (1996) incorporates the age-at-maturity. Body mass-at-age was estimated using the daily size and growth information from Strawn (1958) and Azzarello (1990). Total length was converted to weight from a length-weight relationship (J.K. Carlson, unpublished data). Fecundity was the mean number of male offspring (assuming a 1:1 ratio of males to females) per year (Masonjones and Lewis, 2000). Maximum age for the dwarf seahorse was assumed 2 years.

A life table based upon the Euler–Lotka equation was created in Microsoft® Excel and converted to a post-breeding Leslie matrix for male dwarf seahorse using PopTools (Hood, 2010). Following Caswell (2001), in the matrix formulation the rate of population increase (λ) is equal to the dominant eigenvalue of the Leslie matrix. The developed Leslie matrix model was input to a commercially available software package (RAMAS Metapopulation 6.0; Akçakaya and Root, 2013) to project and examine population responses to conditions set in the models. This model implements a standard Leslie matrix (L) that provided age-specific inputs of fecundity (F_x) and survival (S_x).

The population size (specified as a vector of abundance by age) from one time step (N_t) to the next (N_{t+1}) is given by:

$$N_{t+1} = L_t N_t.$$

Initial population size estimates were developed for the following areas: Cedar Key, Tampa Bay, Charlotte Harbor, Florida Bay, and Northern Indian River Lagoon (Figure 1). These areas were chosen based on Florida's Fish and Wildlife Conservation Commission Fishery-Independent Monitoring (FWC-FIM) survey domains. While other survey domains exists for the FWC-FIM survey (i.e., Apalachicola Bay and Northeast Florida), surveys in these areas captured few dwarf seahorses (Apalachicola Bay) or no dwarf seahorses (Northeast Florida). To develop estimates of initial population size, we used several estimates of population density (m^2) in Tampa Bay, Florida Bay, southwest Florida, Biscayne Bay and north central Florida from

Masonjones et al. (2010) and various personal communications with researchers from National Marine Fisheries Service, Florida Fish and Wildlife Conservation Commission, and United States Geological Survey (Table 2). To determine an initial density estimate, the dataset of values was bootstrapped 10,000 times to yield 5, 10, 25, 50, and 75% quantiles. Dwarf seahorse abundance is generally greatest in areas with higher density of seagrass blades and higher seagrass canopy (e.g., length of seagrass blades) (Lourie et al., 2004). Due to the fact the seagrass density and overall meadow health in Florida is highly variable, for projections, we used a conservative estimate of initial population size by applying the 5% and 10% quantile dwarf seahorse density estimates multiplied by seagrass density in nearshore waters (Yarbro and Carlson, 2016). Based on FWC-FIM surveys, dwarf seahorse relative abundance is higher in Tampa Bay, Charlotte Harbor, and Florida Bay and less abundant in Cedar Key and Northern Indian River Lagoon. Thus, we applied the density estimate from the 10% quantile (0.0030 $N{\cdot}m^{-2}$) for Tampa Bay, Charlotte Harbor and Florida Bay and the 5% quantile (0.0009 $N{\cdot}m^{-2}$) for Cedar Key and Northern Indian River Lagoon.

While there is very little information on the relationship of density dependence for seahorses (Foster and Vincent, 2004; Curtis and Vincent, 2008) due to the likeliness that individuals will compete for resources at high densities, we assumed density dependence followed a Beverton-Holt stock recruitment relationship:

$$R_t = \frac{R_{max} * K}{R_{max} * N_t - N_t + K}$$

Where R_t is the population growth rate at time t, R_{max} is the maximum population increase rate, N_t is the abundance vector at time t, and K is the carrying capacity. RAMAS models density dependence by modifying the select matrix elements at each time step so that the dominant eigenvalue (λ) of the matrix is equal to the growth rate given by the Beverton-Holt stock recruitment relationship. The modification consists of multiplying the matrix elements with a variable m. The value of m depends on the abundance of the population at time step t and is selected to result in the growth rate given by the stock recruitment relationship. The calculation of m given R_t required an iterative algorithm that finds the eigenvalue of the stage matrix at each iterative step. For the population at each time step, RAMAS calculated the parameters of a function that gave *m* as a function of the population size (*N)*. At each time step of each replication, RAMAS used this function to calculate *m*, and multiplied the vital rates by *m*. The eigenvalue of the matrix λ, becomes 1-λ. Further details on the sequence of calculations carried out by RAMAS during each simulation are provided in Akçakaya and Root (2013). As there is little information on the intrinsic rates of increase and R_{max} for seahorses, we assumed R_{max} was equal to the dominant eigenvalue of the Leslie matrix.

Scenarios

Retrospective Analysis

Scenarios were developed to estimate the status of dwarf seahorse among the five areas (Table 3). Scenarios were initiated at the earliest time data were available on the coverage of the seagrass canopy from Yarbro and Carlson (2016). For fishing mortality, we simulated the various levels of harvest on dwarf seahorse by the Marine Life Industry from the Florida Fish and Wildlife Conservation Commission[2]. In the 20 years prior to the development of the current regulations (1990-2009), commercial harvest of dwarf seahorse averaged ~44,000 individuals per year. Average annual commercial harvest was reduced to approximately 17,000 individuals per year, a reduction in annual harvest of approximately 60% per year, from 2010-2016[3]. Landings data is reported by region (e.g., Everglades to Miami, Florida), thus we modeled the reported harvest levels of dwarf seahorse for the area that corresponded to the Florida Fish and Wildlife Conservation Commission data (Table 1).

Dwarf seahorses are assumed to be caught as bycatch in Florida shrimp bait fishery. While the use of any net with a mesh area exceeding 46.5 m^2 has been prohibited in nearshore and inshore waters of Florida (Florida 68B- 4.0081(3)(e)), there is a shrimp bait fishery that operates within these boundaries. Baum et al. (2003) analyzed bycatch of the lined seahorse (*Hippocampus erectus*) in the shrimp bait trawl fishery and estimated about 72,000 seahorses were incidentally caught per year. Baum et al. (2003) reported only two dwarf seahorse were

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[2] Florida Fish and Wildlife Conservation Commission. 2016. A history of Florida's management of the dwarf seahorse (*Hippocampus zosterae*). Unpublished report presented to National Marine Fisheries Service, St. Petersburg, Florida.

captured as part of this study; using the ratio of dwarf seahorse caught to lined seahorse caught, it estimated that 157 are incidentally caught per year. Trawling is prohibited within Biscayne National Park, Dry Tortugas National Park, and Everglades National Park, and from Ecological Reserves, Sanctuary Preservation Areas, and Special-use Research Only areas within the Florida Keys National Marine Sanctuary. Thus, we did not consider any bycatch from Florida Bay.

Loss or gain in seagrass habitat (i.e., carrying capacity) was modeled using the documented historic coverage of seagrass in each respective system from Yarbro and Carlson (2016). Data for years when surveys was not conducted was estimated using the slope of the trend in seagrass coverage calculated from Yarbro and Carlson (2016). Carrying capacity (K) was assumed to be the maximum density of dwarf seahorse that seagrass habitat could support, given a seagrass habitat of high blade density and canopy. Still applying a conservative approach, the 25% quantile estimate ($0.02\ N \cdot m^{-2}$) from the bootstrapped dataset on dwarf seahorse density was applied to the area, as previously described.

Given dwarf seahorse have limited movements and would likely experience some mortality due to cold temperatures, we examined the effects of cold exposure. We modeled acute cold mortality events (minimum air temperatures below 30° C for more than 3 days; *sensu* Gilmore et al., 1978, Rehage et al., 2010) using data (1992-2016, depending on the area being modeled) from the Florida State University-Office of the State Climatologist (http://climatecenter.fsu.edu/climate-data-access-tools/downloadable-data). No data are available on thermal tolerance of dwarf seahorse. Although Mascaró et al. (2016) found survival was not affected when juvenile *Hippocampus erectus* (a sympatric species to dwarf seahorse) were exposed to acute temperatures (i.e., 7°C), we adopted a conservative approach by assuming a theoretical mortality level of 30% for all individuals in that area during the year of the cold event based on the survivorship rates for other species of seahorse subjected to various temperature regimes (Lin et al., 2006; Martinez-Cardenas and Purser, 2011; Planas et al., 2012).

Acute mortality was modeled for events associated with HABs. HABs were considered for red tide (*Karenia brevis*), blue-green tide (*Chlorophyta*), and brown tide (*Aureoumbra lagunensis*) events. The frequency of these events (1992-2016, depending on the area being modeled) was obtained from data from the Florida Fish and Wildlife Conservation Commission website (http://myfwc.com/research/redtide/monitoring/database) and Yarbro and Carlson, (2016). Mortality was estimated considering the area affected by the HAB. As we do not have accurate data on the extent of the HAB, we assumed a theoretical acute mortality of 25% based on the fact that in most cases in southwest Florida, HABs are generally found in the coastal zone outside the bays where dwarf seahorse are found. The exception was when the extent of the HAB was extreme and better documented (e.g., the 2011 "superbloom" in Northern Indian River Lagoon; Yarbro and Carlson, 2016) and acute mortality was increased to 50%.

Stochasticity was incorporated into the model as new abundance vectors (N_{t+1}) by randomly drawing values specified in the Leslie matrix. At each time step, a random variable is drawn for each vital rate (i.e., survival and fecundity) based on a lognormal distribution and the standard deviation assigned to each vital rate in the matrix. Given the uncertainty in estimates of survivorship, standard deviation was based on an assumed coefficient of variation of 30%. Each time step was replicated 1,000 times. RAMAS introduces variation in carrying capacity (K) by randomly sampling a single deviate at each time step based on the estimated standard deviation given for K. Measurement error in the initial population size and harvest levels (i.e., fishing mortality) was set by specifying a coefficient of variation (CV) of 0.3.

Population Projection Scenarios

Future scenarios were developed to test the effects of the most likely threats to the dwarf seahorse in Florida (Table 4). Again, applying a conservative approach, initial population sizes were estimated based on the same density estimates calculated for the retrospective analysis or the most recent estimate of coverage of seagrass habitat by area (Yarbro and Carlson, 2016). As the harvest of dwarf seahorse by the Marine Life Industry has been limited by regulation, the greatest threats to future seahorse populations include the loss of seagrass habitat and increased HABs, which can cause acute mortality. We developed optimistic (no change from baseline and increased seagrass coverage) and pessimistic scenarios based on increased rates of mortality, loss of seagrass habitat, and likelihood of HABs increasing (Table 3).

Although recent information on the status of seagrass indicates that most of the areas modeled herein had either stable or increasing seagrass habitat (Yarbro and Carlson, 2016), we modeled a theoretical decrease in available habitat (loss in carrying capacity) of 25% and 50% over the projection. When considering HABs, we examined the current rate of HABs for each area over the last 10 years. We then increased that rate over the simulation period. For example, in Northern Indian River Lagoon a harmful algal bloom has occurred six times over the last 10 years (2007, 2010-2013, 2016). For these projections, we increased the probability of an HAB occurring to 0.7 and 0.8 every year.

The time period for the population projection was determined following IUCN Red List Categories and Criteria when classifying species at a risk of extinction (Version 3.1: IUCN, 2012). Threats likely to occur in the "future" are evaluated using a time frame of three generations or ten years, whichever is longer (not exceeding 100 years in the future). The current generation time of the dwarf seahorse is 1.24 years, thus 3 generations=3.72 years. Therefore, considering the IUCN Red List criteria for future threats, we projected the population forward for 10 years.

Results

In the retrospective analysis**,** population size was estimated to range from about 15,388 males in Cedar Key to 2,081,036 males in Florida Bay. Similarly, in the future projections, initial population size was greatest in Florida Bay and lowest in Cedar Key (Table 2). Based on current life history parameters, the estimate of population growth (λ) for dwarf seahorse was 1.46 yr^{-1}. The intrinsic rate of population increase (r) was 0.38 yr^{-1}, calculated assuming the finite rate of population increase (λ) is analogous to $e^r=\lambda$. The estimates of the expected number of replacements (R_o) and generation time (T) were 1.60 and 1.24 years, respectively.

Retrospective Simulations

The simulated population trajectories from the retrospective scenarios ranged from increases in abundance (Tampa Bay, Charlotte Harbor, Florida Bay), to a relative stable trend (Cedar Key), and a reduction in population size (Northern Indian River Lagoon) (Figure 2-6). The highest proportional change from 1996 to 2016 was in Tampa Bay where the population increased by 7 times its initial abundance (Figure 7). Florida Bay and Charlotte Harbor also increased by 3-5 times since 1992 and 1997, respectively. There were minimal changes in abundance predicted for Cedar Key since 2001.

Since most areas are experiencing stable or increasing trends in seagrass, populations in each area were most affected by the occurrence of HABs. The exception was Cedar Key whose population increase was most affected by cold temperature but the population remained relatively stable through the simulation (Figure 2). The Northern Indian River Lagoon population was found to increase from 1992 to 2007 when a severe HAB caused the population to decline rapidly; subsequent HABs in 2010-2013, combined with a reduction in carrying capacity, prevented the population from recovering (Figure 6). Similarly, the Florida Bay population declined following HABs in 2007 and 2009, but was able to recover due to the extensive remaining areas of seagrass (Figure 5). The Tampa Bay and Charlotte Harbor populations were also observed to respond positively to the reduction in commercial harvest of seahorse by the Marine Life Industry in 2010 (Figure 3-4).

Future Simulations

For Cedar Key, all projection scenarios resulted in negligible (1% increase or decrease) population changes. All scenarios fluctuated around the initial biomass with some slight increases or decreases depending on the scenario (Figure 8). Cold temperature events, which occurred annually, were a significant source of mortality preventing the population from growing much larger than the initial abundance (Figure 8; Scenarios 1-5).

All the scenarios for Tampa Bay (Figure 9; Scenarios 7-10), Charlotte Harbor (Figure 10, Scenarios 13-17), and Florida Bay (Figure 11, Scenarios 19-23) showed some population increase although in the most pessimistic scenarios the increase was negligible or slightly decreasing (Scenarios 12, 18, 24). Under the two optimistic scenarios, the mean population size grew rapidly, increasing up to 3.0-3.8 times from the initial abundance in 10 years (Figure 12). For the pessimistic scenarios, the mean population still increased about 15% from the initial population size. However, there was considerable variability in the pessimistic scenarios with the lower standard deviation values for some scenarios declining up to 60% from the initial population size in Charlotte Harbor. Florida Bay contained the most robust population with any decline in abundance negligible.

The five Northern Indian River Lagoon scenarios were the most pessimistic of all areas. One scenario resulted in the population growing about 2 times from initial abundance, two scenarios showed negligible population increases, and two scenarios showed a mean population decline of >80,000 individuals over 10 years with Scenario 30 resulting in a decline to 49 individuals (Figure 13). The lower standard deviation values for scenarios 24 and 25 resulted in the extirpation of dwarf seahorses within 5 years of the projection in the Northern Indian River Lagoon system.

Discussion

Seahorse population sizes are generally considered low, making them susceptible to harvest. Because of the variation in the quality and quantity of seagrass in Florida, we used a more conservative density estimate for the dwarf seahorse in this study when compared to other populations of seahorse (Foster and Vincent, 2004). However, dwarf seahorse populations still numbered in the millions, especially in Florida Bay. The high population numbers of the dwarf seahorse are likely linked to the expansive seagrass habitat throughout Florida, similar to evaluations of abundance of other seahorse species. The most recent mapping data indicates there are about 10,036 km^2 of seagrass in nearshore Florida waters, with most located in southern Florida (Yarbro and Carlson, 2016). Curtis and Vincent (2008) also determined a large population size (~2,000,000 animals) for European long-snouted seahorse in the Ria Formosa, Portugal, and determined the high population estimate was due to the unusually high productivity and expansive seagrass habitat of this area (total lagoon area is ~170 km^2).

The life history characteristics of the dwarf seahorse (i.e., early age at maturity, rapid growth, and high fecundity) indicate that this species has a relatively high intrinsic rate of population increase (R_{max}). Unfortunately, very little demographic modeling has been conducted for any seahorse species to provide a comparison. In a PVA for European long-snouted seahorses, Curtis (2004) estimated R_{max}=6.8 yr^{-1} based on a regression of known R_{max} values to body size for 43 exploited marine fish stocks (Denney et al., 2002). Using the regression of Denney et al. (2002), R_{max} for dwarf seahorse is 9.4 yr^{-1}. However, this value is high and out of the range of values observed for other exploited fishes (Myers et al., 1999). While it is likely that the dwarf seahorse has high productivity, the demographic approach to determine population increase does not take into account the consequences of this species' reproductive and social behaviors. Further research on how monogamy, limited dispersal, and mating behavior, as well as to what extent these behaviors effect population growth rates under low population sizes (e.g., Allee effects), needs to be explored.

Dwarf seahorses have limited active dispersal (Vincent, 1996) largely due to low mobility and high site fidelity to small home ranges (Fedrizzi et al., 2015). However, dwarf seahorse have been observed clinging to drift microalgae in sample nets. Rafting could also serve as a method of dispersal for dwarf seahorses (Masonjones et al., 2010) as noted for other seahorses (Luzzatto et al., 2013). Fedrizzi et al. (2015) found significant population structuring with strong evidence for a distinct population in the Florida Panhandle, two recognizable sub-populations in the Florida Keys, and a potential fourth sub-population at Big Pine Key. Dwarf seahorse from the Northern Indian River Lagoon was not delineated as a discrete population due to small sample size and lack of consistency in relationship to the other populations. Despite overall significant population structuring, Fedrizzi et al. (2015) observed evidence of some gene flow for dwarf seahorse between sampled locations, with the exception of the Florida Panhandle. While it is likely there are exchanges of individuals in those areas closest to each other, given the limited information available on dispersal rates for dwarf seahorse, we considered the dwarf seahorse to be found in five separate, though not genetically distinct, populations.

Based on the retrospective analysis, the current status of the populations in 4 of the 5 areas in Florida appears to be healthy. Abundance was found to be relatively constant (Cedar Key), or increasing from the beginning of the simulation (Tampa Bay). These trends tended to follow the seagrass coverage in these bays, which has also been stable or increasing (Yarbro and Carlson, 2016). According to data from the FWC-FIM, dwarf seahorse is most abundant in Charlotte Harbor, Tampa Bay, and Florida Bay[3]. Relative abundance from some of these surveys was similar to the retrospective simulations from the population viability model. While abundance is lower in Cedar Key, the general trend since 1996 has been stable. In Florida Bay from 2006-2009, FWC-FIM data show an increasing trend in relative abundance. The U.S. Geological Survey data from 2005-2011 in Florida Bay also suggest the relative abundance of dwarf seahorse is stable to increasing (M. Robblee, USGS, pers. comm. to K.L. Smith). A decrease in abundance occurred from 2007-2011 in the PVA retrospective analysis from Florida Bay but since 2012 there have been increases, but not to historical highs observed in the early 2000s. In contrast to increases in abundance predicted by the PVA simulations, data from FWC-FIM surveys for Tampa Bay and Charlotte Harbor show a relative declining trend from 1996-2010; however, this trend begins to increase in 2011[6]. The FWC-FIM measured declines are somewhat unexpected given seagrass coverage has been increasing in these areas since 1996 (Yarbro and Carlson, 2016) and dwarf seahorse population density in higher quality habitat was

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[3] Florida Fish and Wildlife Conservation Commission. 2016. A history of Florida's management of the dwarf seahorse (*Hippocampus zosterae*). Unpublished report presented to National Marine Fisheries Service, St. Petersburg, Florida.

estimated at 0.18 seahorse m^2 in Tampa Bay during the wet season (Masonjones et al., 2010). The declining trend could be a reflection of the areas sampled by the FWC-FIM surveys (lower quality habitat) and the impact of the aquarium harvest by the Marine Life Industry. It is worthy to note that both the retrospective PVA simulations and the relative abundance trends from FWC-FIM surveys responded positively to the reduction in annual harvest of approximately 60% per year by the Marine Life Industry.

The retrospective scenario for Northern Indian River Lagoon was the most pessimistic, which could indicate the status of the dwarf seahorse in this system is poor. While the population in the Northern Indian River Lagoon increased from 1992 to 2008, beginning in 2009 the population to decline rapidly. The FWC-FIM survey data also show that dwarf seahorse relative abundance has been reduced or absent in this area since 2010[4]. The decline in abundance from both the PVA and FWC-FIM survey is correlated with a HAB that began in 2011 and was larger than any past events in terms of geographic scale, bloom intensity, duration, and seagrass loss (St. Johns Water Management District et al., 2012). The 2011 seagrass coverage in the Northern Indian River Lagoon estuary was reduced by 60% compared to 2010[3]. Similarly, FWC's Seagrass Integrated Mapping and Monitoring (SIMM) program reported seagrass coverage declined in the southern Indian River Lagoon by almost 2,000 acres between 2009 and 2011 (Robbins et al., 2016). These declines were followed by increases in seagrass coverage throughout the Northern Indian River Lagoon between 2011 and 2013 (Robbins et al., 2016, Morris et al., 2016). Despite these increases, the population does not to appear to have recovered in either FWC-FIM surveys or the retrospective PVA simulation.

Simulations indicated the most significant factor affecting dwarf seahorse in the future would be HABs combined with the associated loss of habitat. Large-scale harmful algal blooms have occurred in dwarf seahorse habitat in Florida Bay, Biscayne Bay, and the Northern Indian River Lagoon systems, resulting in substantial habitat loss and mortality. In the 1990s, a large HAB resulted in extensive seagrass die-offs in Florida Bay. However, due to Florida Bay being a core area of high abundance and containing significant seagrass coverage, dwarf seahorse

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[4] Florida Fish and Wildlife Conservation Commission. 2016. A history of Florida's management of the dwarf seahorse (Hippocampus zosterae). Unpublished report presented to National Marine Fisheries Service, St. Petersburg, Florida.

population in there seems to have been able to withstand the loss. Thus, areas with lower baseline densities of dwarf seahorse populations (i.e., northern portions of their range) may not be able to withstand harmful algal blooms that result in significant habitat loss.

The population of the dwarf seahorse within Cedar Key, Tampa Bay, Charlotte Harbor and Florida Bay appears to be healthy. Despite some commercial harvest and removal as bycatch occurring within their range, the retrospective analysis suggests populations in these areas are stable or increasing. Protections afforded by Everglades National Park, seasonal closures to trawling along the west coast of Florida, and reductions in aquarium harvest will further help to conserve the species. However, in areas such as Northern Indian River Lagoon, frequent and large HABs are having a significant impact on the species. Although the dwarf seahorse is highly productive and can sustain some level of harvest, the combined impact of habitat loss and mortality will hinder its ability to recover in these areas. In addition, while the trend in seagrass coverage is currently reported to be positive in Tampa Bay, Charlotte Harbor, and Florida Bay, any loss in seagrass habitats or increases in HABs will likely adversely affect the status of dwarf seahorse in these areas. Future research on the effects HABs have on the seagrass canopy and its ability to recover following these events needs to be explored.

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Table 1. Life history and baseline parameters used in the development of the dwarf seahorse *Hippocampus zosterae* population viability model.

Parameter	Value	Source/Method
Age-at-maturity:	3 months	Wilson and Vincent (2000)
Maximum age:	2+ yrs	Lourie et al. (2004)
Fecundity:	130.2 males yr^{-1}	Strawn (1958) Masonjones and Lewis (2000)
Survivorship:		
Age 0	0.00888 yr^{-1}	Multiple indirect methods
Age 1	0.383 yr^{-1}	Masonjones et al. (2010)
Age 2	0.383 yr^{-1}	Masonjones et al. (2010)
	a= 0.000014 b=·4724	This study
Length-weight relationship:		
Von Bertalannfy growth parameters:	L_∞ (cm)= 37.2 K=3.24 t_0-0.1001	This study
R_{max}:	1.49 yr^{-1}	This study
Initial population size:		
Retrospective analysis		
Cedar Key	15,388 males	This study
Tampa Bay	138,457 males	This study
Charlotte Harbor	359,703 males	This study
Florida Bay	2,081,036 males	This study
North Indian River Lagoon	111,019 males	This study
Future projections		
Cedar Key	15,621 males	This study
Tampa Bay	215,825 males	This study
Charlotte Harbor	390,313 males	This study
Florida Bay	1,653,780 males	This study
North Indian River Lagoon	86,742 males	This study

Table 2. Density estimates used in estimating initial population size for dwarf seahorse populations in Florida.

Mean density (m^2)	Range (m^2)	Area	Source
0.363	0.178-0.599	Biscayne Bay, FL	Mike Robblee, USGS, pers. comm. to Kelcee Smith
0.030	-	Florida Bay, FL	Sogard et al (1989)
0.217	0.099-0.412	Florida Bay, FL	Mike Robblee, USGS, pers. comm. to Kelcee Smith
0.180	-	Florida Bay, FL	Sogard et al (1990)
0.260	-	Florida Bay, FL	Sogard et al (1988)
0.015	0.00-0.075	Southwest Florida coast	Mike Robblee, USGS, pers. comm. to Kelcee Smith
0.003	0.001-0.007	St. Andrews Bay, FL	Stacey Harter, pers. comm. to John Carlson
0.080	0.020-0.180	Tampa Bay, FL	Masonjones et al. (2010)

Table 3. Summary of the retrospective scenarios for the dwarf seahorse *Hippocampus zosterae* population viability analysis.

Population	Starting year	Seagrass trend	Aquarium commercial harvest (mean number of individuals)	Harmful algal bloom mortality	Shrimp trawl bycatch	Cold event mortality
Cedar Key	2001	Stable	413 yr^{-1} (2001-2010) 131 yr^{-1} (2011-2016)	None	157 yr^{-1}	0.3
Tampa Bay	1996	Increasing	20,703 yr^{-1} (1996-2010) 2,097 yr^{-1} (2011-2016)	0.25	157 yr^{-1}	0.3
Charlotte Harbor	1999	Increasing	24,514 yr^{-1} (1999-2010) 2,096 yr^{-1} (2011-2016)	0.25	157 yr^{-1}	0.3
Florida Bay	1992	Stable	15,668 yr^{-1} (1992-2010) 14,780 yr^{-1} (2011-2016)	0.25	None	0.3
Northern Indian River Lagoon	1992	Decreasing	209 yr^{-1} (1992-2010) 190 yr^{-1} (2011-2016)	0.25	157 yr^{-1}	0.3

Table 4. Summary of the population projections for the dwarf seahorse *Hippocampus zosterae* population viability analysis.

Scenario	Area	Seagrass trend	Commercial harvest (seahorse/year)	Bycatch (seahorse/year)	Probability of HAB	HAB mortality	Probability of cold event
1	Cedar Key	Stable	131	157	0.0	No	1.0
2	Cedar Key	Increasing by 10%	131	157	0.0	No	1.0
3	Cedar Key	Decreasing by 25%	131	157	0.1	25%	1.0
4	Cedar Key	Decreasing by 25%	131	157	0.1	35%	1.0
5	Cedar Key	Decreasing by 50%	131	157	0.1	35%	1.0
6	Cedar Key	Decreasing by 50%	131	157	0.1	50%	1.0
7	Tampa Bay	Stable	2,097	157	0.2	25%	0.3
8	Tampa Bay	Increasing by 10%	2,097	157	0.2	25%	0.3
9	Tampa Bay	Decreasing by 25%	2,097	157	0.4	25%	0.3

10	Tampa Bay	Decreasing by 25%	2,097	157	0.4	35%	0.3
11	Tampa Bay	Decreasing by 50%	2,097	157	0.4	35%	0.3
12	Tampa Bay	Decreasing by 50%	2,097	157	0.4	50%	0.3
13	Charlotte Harbor	Stable	2,096	157	0.5	25%	0.2
14	Charlotte Harbor	Increasing by 10%	2,096	157	0.5	25%	0.2
15	Charlotte Harbor	Decreasing by 25%	2,096	157	0.6	25%	0.2
16	Charlotte Harbor	Decreasing by 25%	2,096	157	0.6	35%	0.2
17	Charlotte Harbor	Decreasing by 50%	2,096	157	0.6	35%	0.2
18	Charlotte Harbor	Decreasing by 50%	2,096	157	0.6	50%	0.2
19	Florida Bay	Stable	14,780	None	0.3	25%	0.2

20	Florida Bay	Increasing by 10%	14,780	None	0.3	25%	0.2
21	Florida Bay	Decreasing by 25%	14,780	None	0.5	25%	0.2
22	Florida Bay	Decreasing by 25%	14,780	None	0.5	35%	0.2
23	Florida Bay	Decreasing by 50%	14,780	None	0.5	35%	0.2
24	Florida Bay	Decreasing by 50%	14,780	None	0.5	50%	0.2
25	Northern Indian River Lagoon	Stable	190	157	0.6	25%	0.4
26	Northern Indian River Lagoon	Increasing by 10%	190	157	0.6	25%	0.4
27	Northern Indian River Lagoon	Decreasing by 25%	190	157	0.8	25%	0.4
28	Northern Indian River Lagoon	Decreasing by 25%	190	157	0.8	50%	0.4

29	Northern Indian River Lagoon	Decreasing by 50%	190	157	0.8	50%	0.4
30	Northern Indian River Lagoon	Decreasing by 50%	190	157	0.8	75%	0.4

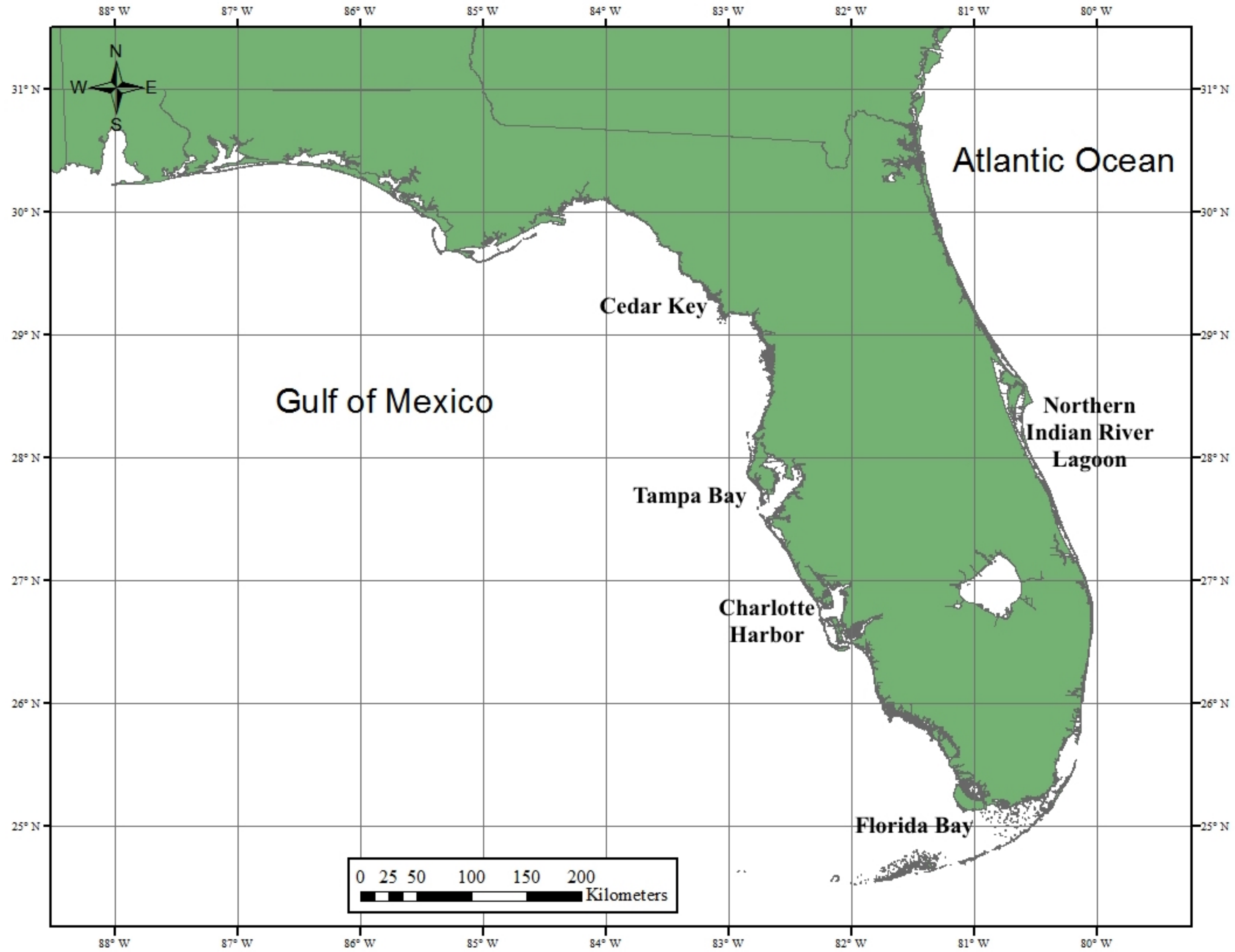

Figure 1. Areas considered for the dwarf seahorse population viability model.

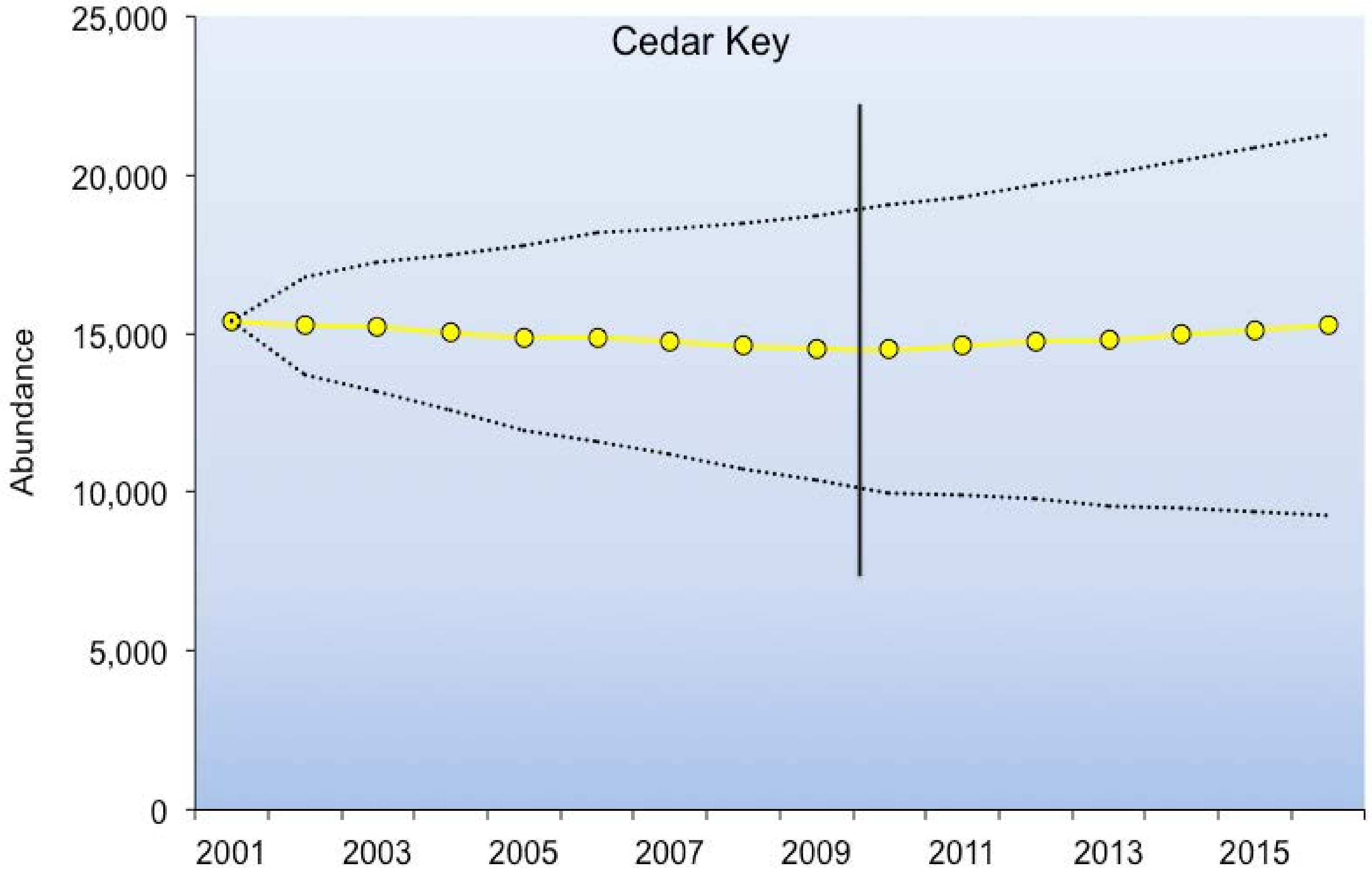

Figure 2. Retrospective simulations of dwarf seahorse in Cedar Key, Florida. Yellow circles represent the mean abundance at time. Dotted lines represent the ±1 standard deviation of the population mean at time *t*. Vertical line indicates the year the regulation for the reduction in annual harvest by the Marine Life Industry was implemented.

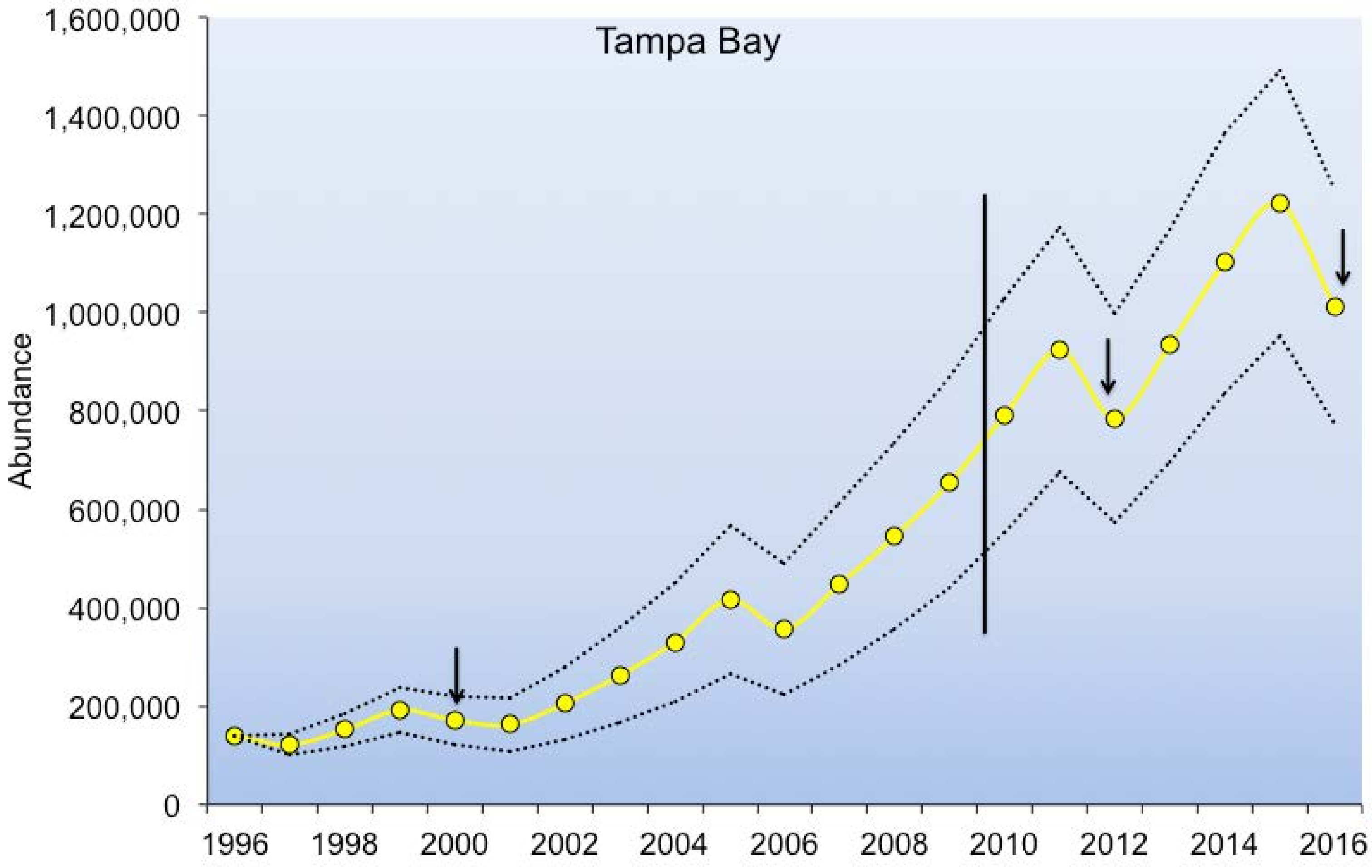

Figure 3. Retrospective simulations of dwarf seahorse in Tampa Bay, Florida. Yellow circles represent the mean abundance at time. Dotted lines represent the ±1 standard deviation of the population mean at time. Vertical line indicates the year the regulation for the reduction in annual harvest by the Marine Life Industry was implemented. Arrows are indicative of the year a harmful algal bloom occurred.

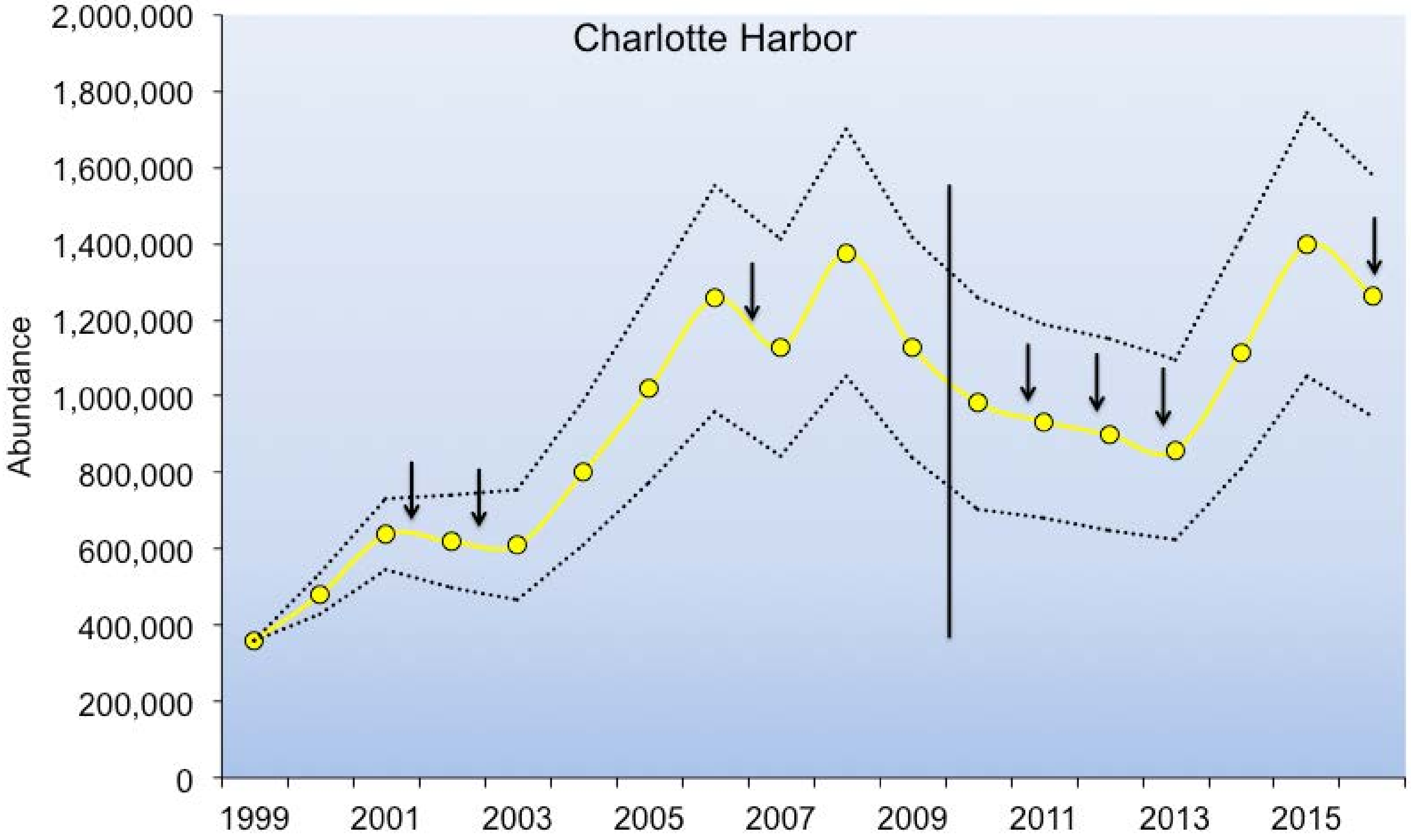

Figure 4. Retrospective simulations of dwarf seahorse in Charlotte Harbor, Florida. Yellow circles represent the mean abundance at time. Dotted lines represent the ±1 standard deviation of the population mean at time. Vertical line indicates the year the regulation for the reduction in annual harvest by the Marine Life Industry was implemented. Arrows are indicative of the year a harmful algal bloom occurred.

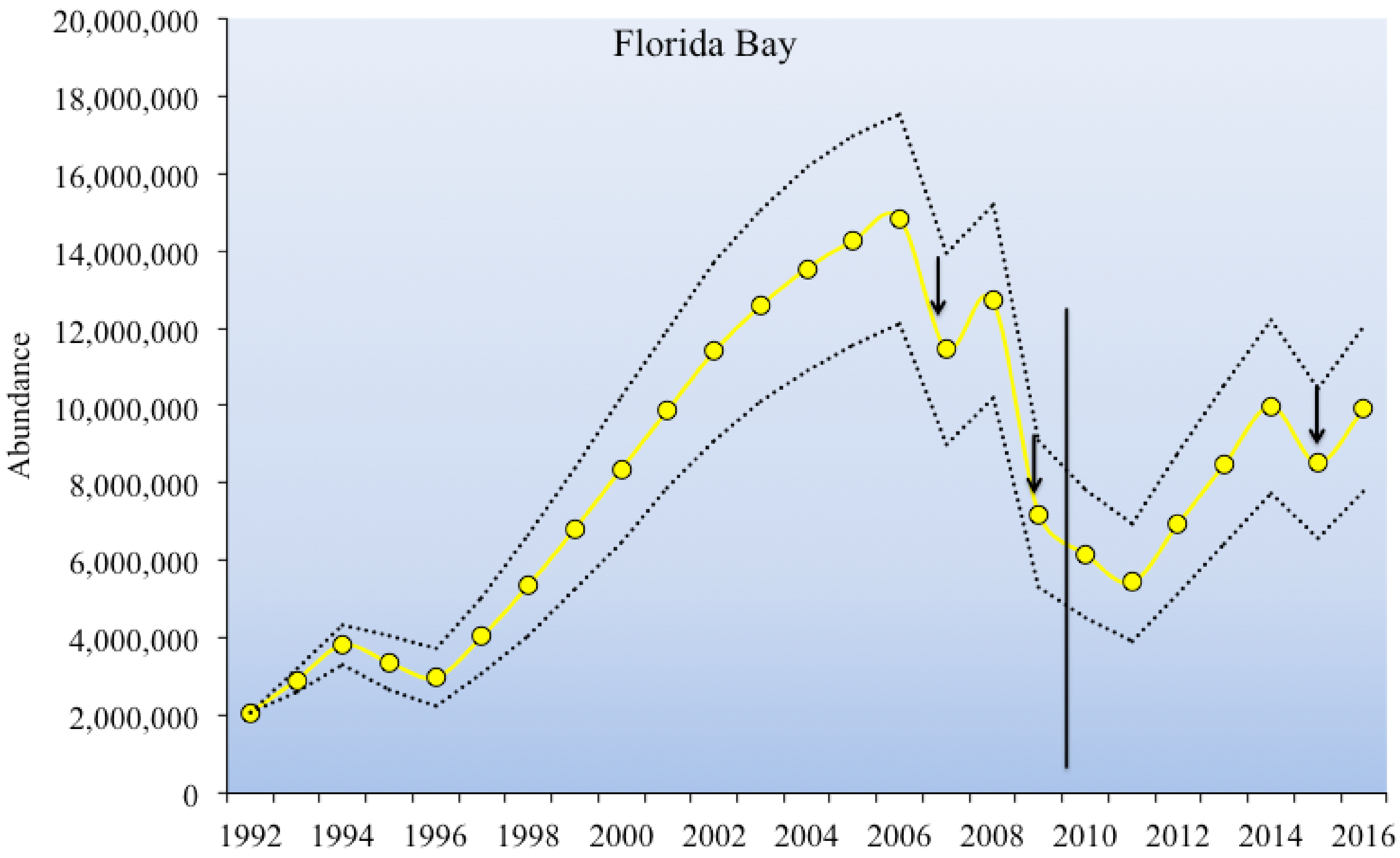

Figure 5. Retrospective simulations of dwarf seahorse in Florida Bay, Florida. Yellow circles represent the mean abundance at time. Dotted lines represent the ±1 standard deviation of the population mean at time. Vertical line indicates the year the regulation for the reduction in annual harvest by the Marine Life Industry was implemented. Arrows are indicative of the year a harmful algal bloom occurred.

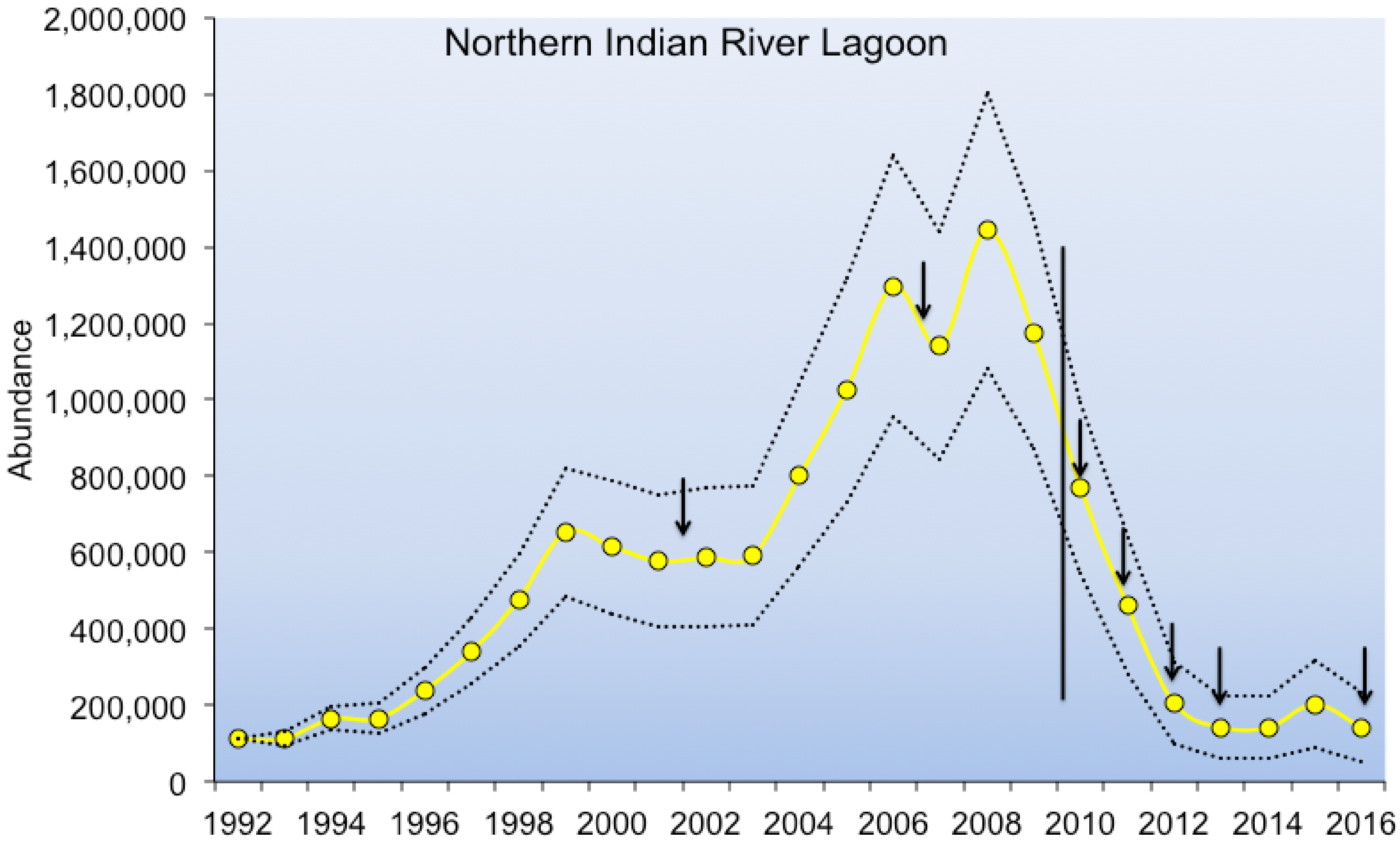

Figure 6. Retrospective simulations of dwarf seahorse in Northern Indian River Lagoon, Florida. Yellow circles represent the mean abundance at time. Dotted lines represent the ±1 standard deviation of the population mean at time t. Vertical line indicates the year the regulation for the reduction in annual harvest by the Marine Life Industry was implemented. Arrows are indicative of the year a harmful algal bloom occurred.

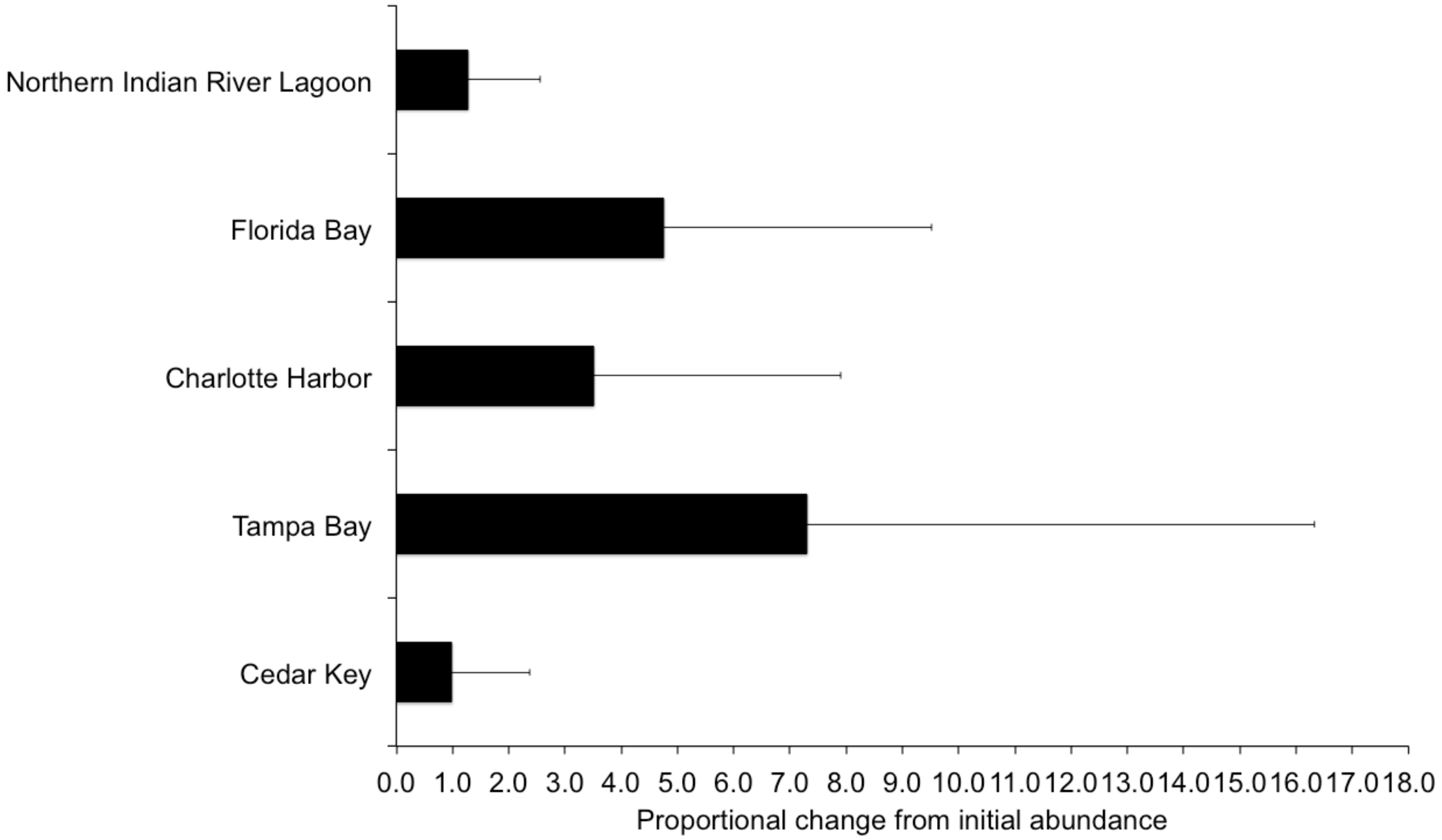

Figure 7. Mean proportional difference from the initial population size to the population size at the end of the scenario for the retrospective analysis. Error bars represent ±1 standard deviation of the mean proportional difference. A value less than 1.0 would indicate the population decreased from initial abundance.

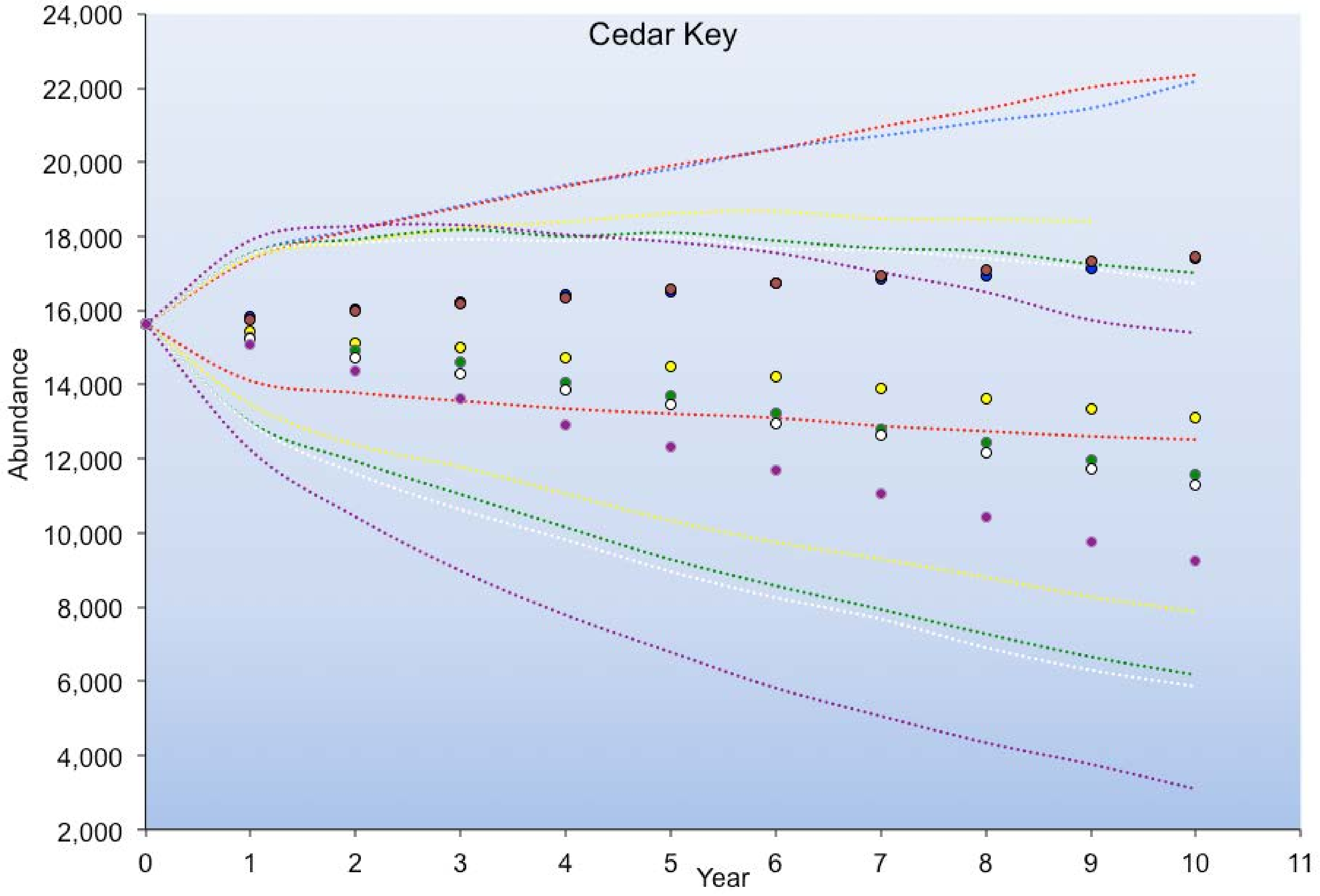

Figure 8. Projections of dwarf seahorse abundance at time t under scenarios exploring optimistic and pessimistic simulations (see Table 3 for details) for Cedar Key, FL. Blue circles = scenarios 1. Red circles = scenarios 2. Yellow circles = scenarios 3. Green circles = scenarios 4. White circles = scenarios 5. Purple circles = scenarios 6. Dotted lines represent ±1 standard deviation of the population mean at time t with the colors corresponding to the mean abundance symbols.

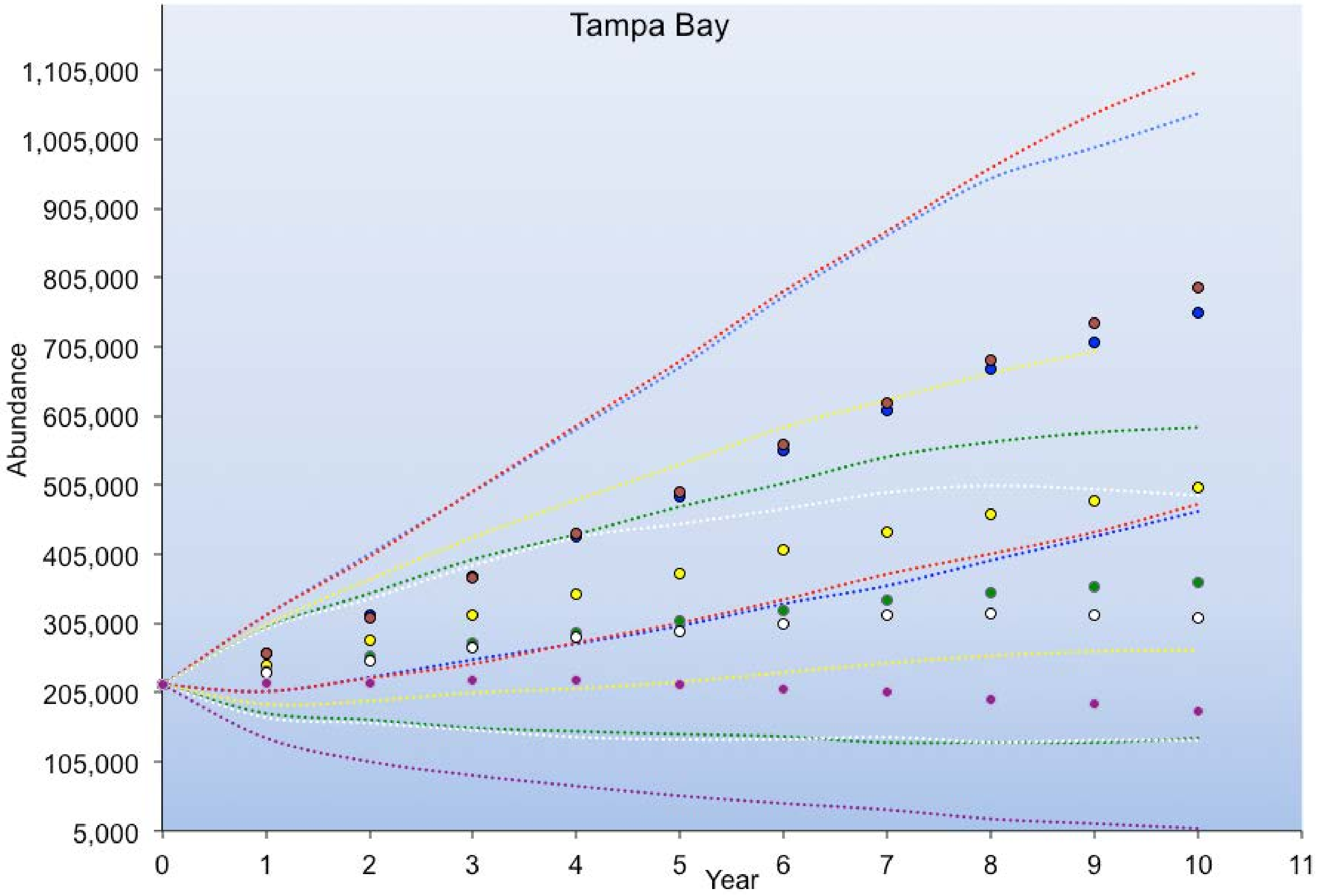

Figure 9. Projections of dwarf seahorse abundance at time t under scenarios exploring optimistic and pessimistic simulations (see Table 3 for details) for Tampa Bay, FL. Blue circles = scenarios 7. Red circles = scenarios 8. Yellow circles = scenarios 9. Green circles = scenarios 10. White circles = scenarios 11. Purple circles = scenarios 12. Dotted lines represent ±1 standard deviation of the population mean at time t with the colors corresponding to the mean abundance symbols.

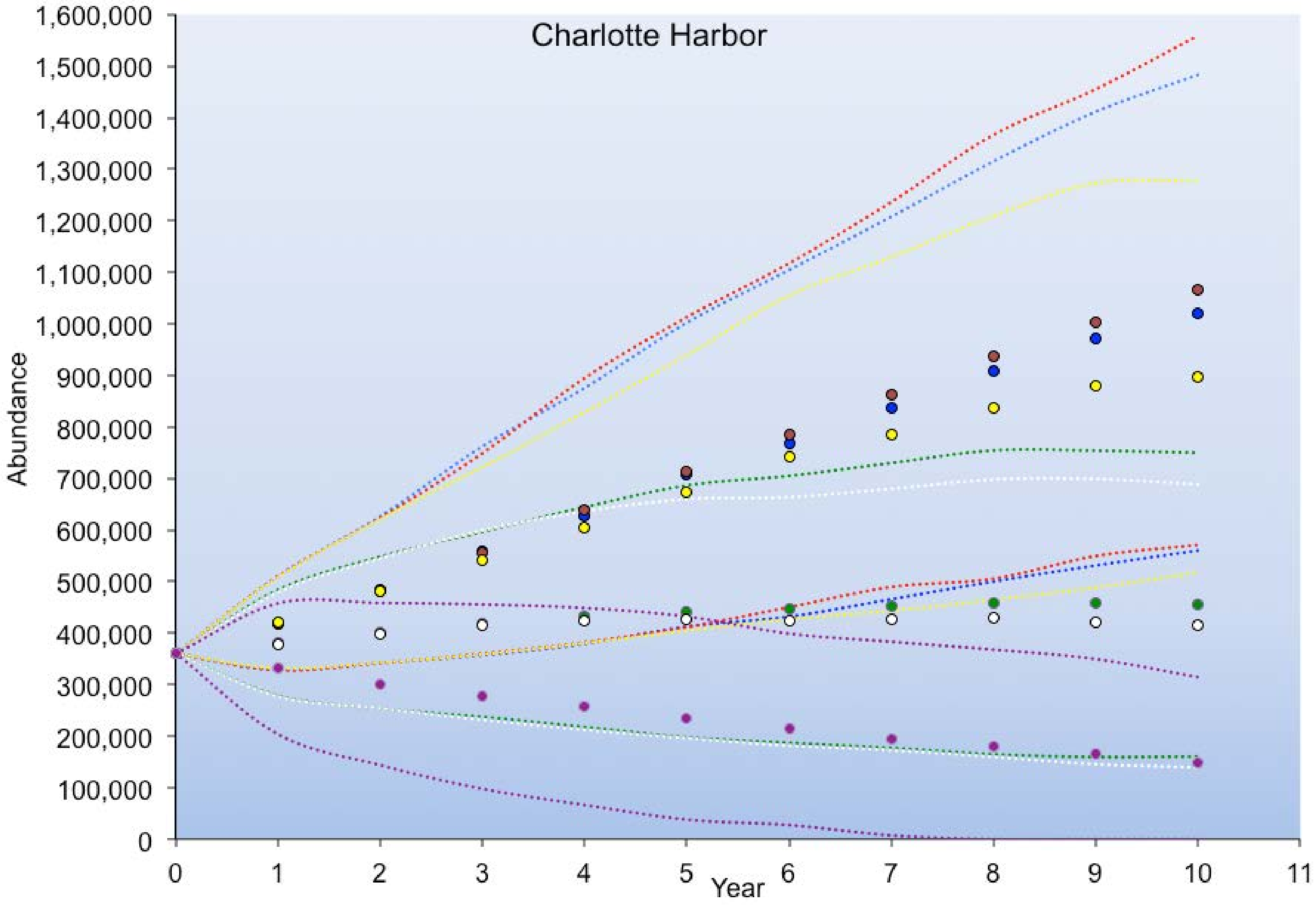

Figure 10. Projections of dwarf seahorse abundance at time t under scenarios exploring optimistic and pessimistic simulations (see Table 3 for details) for Charlotte Harbor, FL. Blue circles = scenarios 13. Red circles = scenarios 14. Yellow circles = scenarios 15. Green circles = scenarios 16. White circles = scenarios 17. Purple circles = scenarios 18. Dotted lines represent ±1 standard deviation of the population mean at time t with the colors corresponding to the mean abundance symbols.

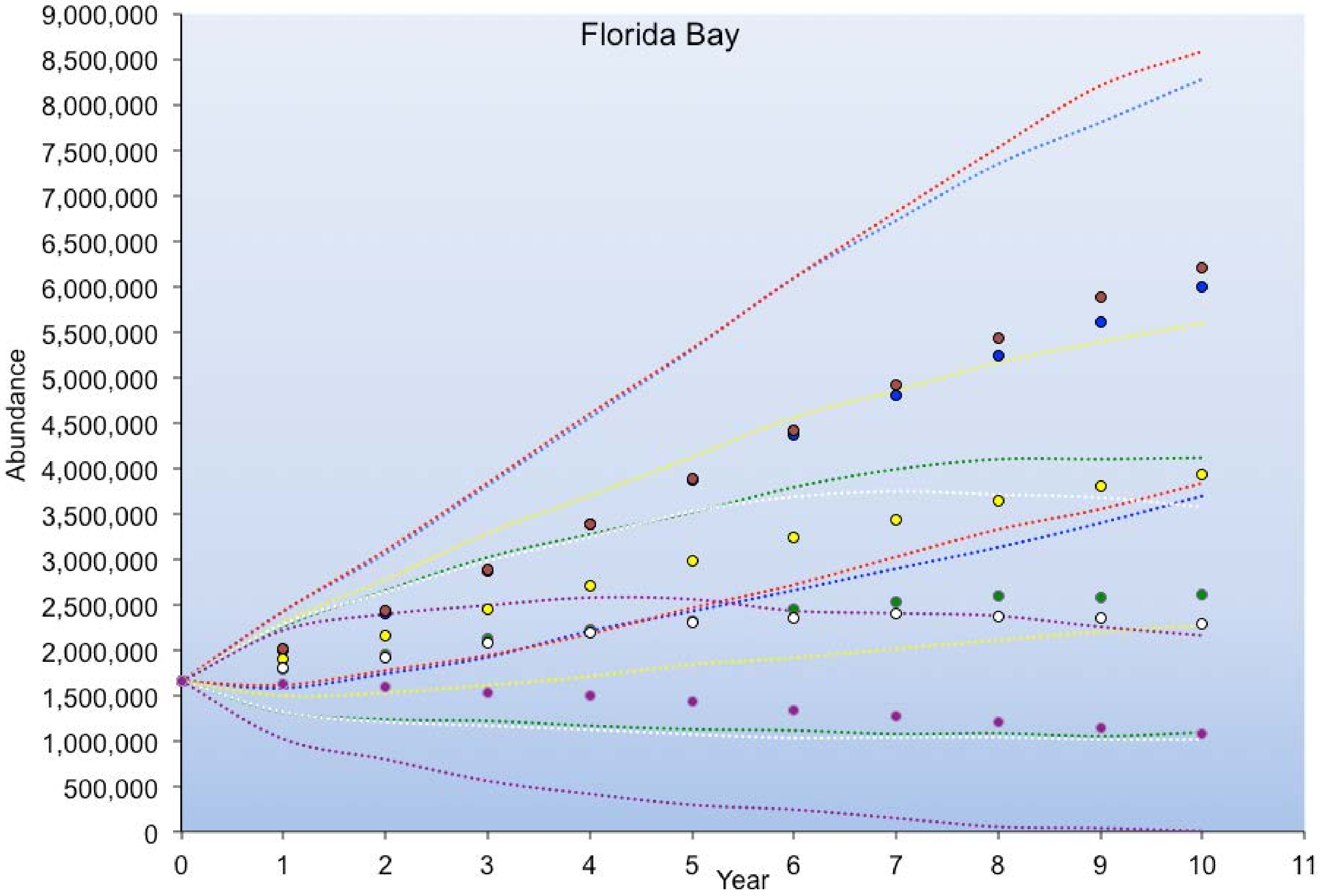

Figure 11. Projections of dwarf seahorse abundance at time t under scenarios exploring optimistic and pessimistic simulations (see Table 3 for details) for Florida Bay, FL. Blue circles = scenarios 19. Red circles = scenarios 20. Yellow circles = scenarios 21. Green circles = scenarios 22. White circles = scenarios 23. Purple circles = scenarios 24. Dotted lines represent ±1 standard deviation of the population mean at time t with the colors corresponding to the mean abundance symbols.

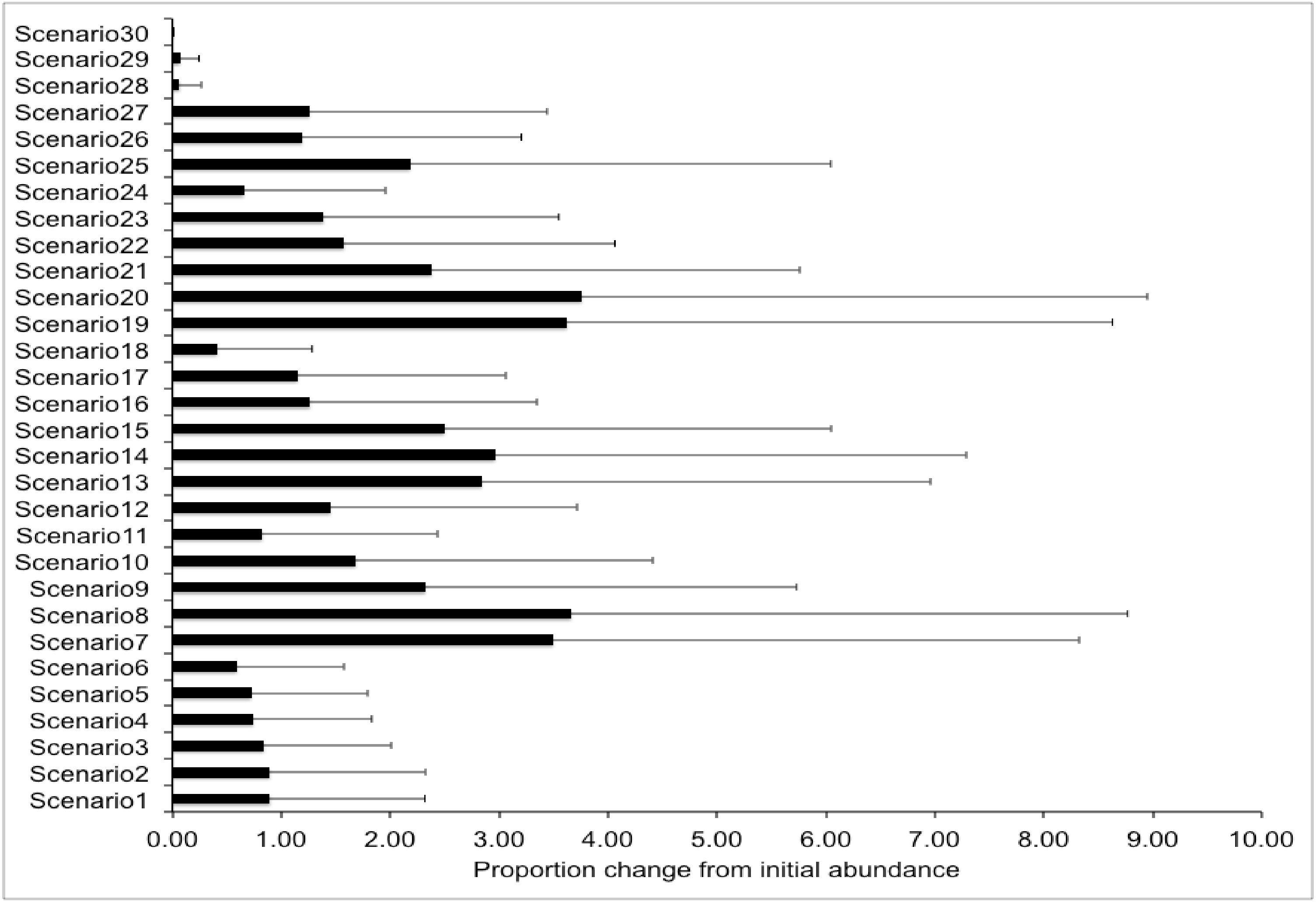

Figure 12. Mean proportional difference from the initial population size to the population size at the end of the scenario for the optimistic and pessimistic simulations. Error bars represent ±1 standard deviation of the mean proportional difference. A value less than 1.0 would indicate the population decreased from initial abundance.

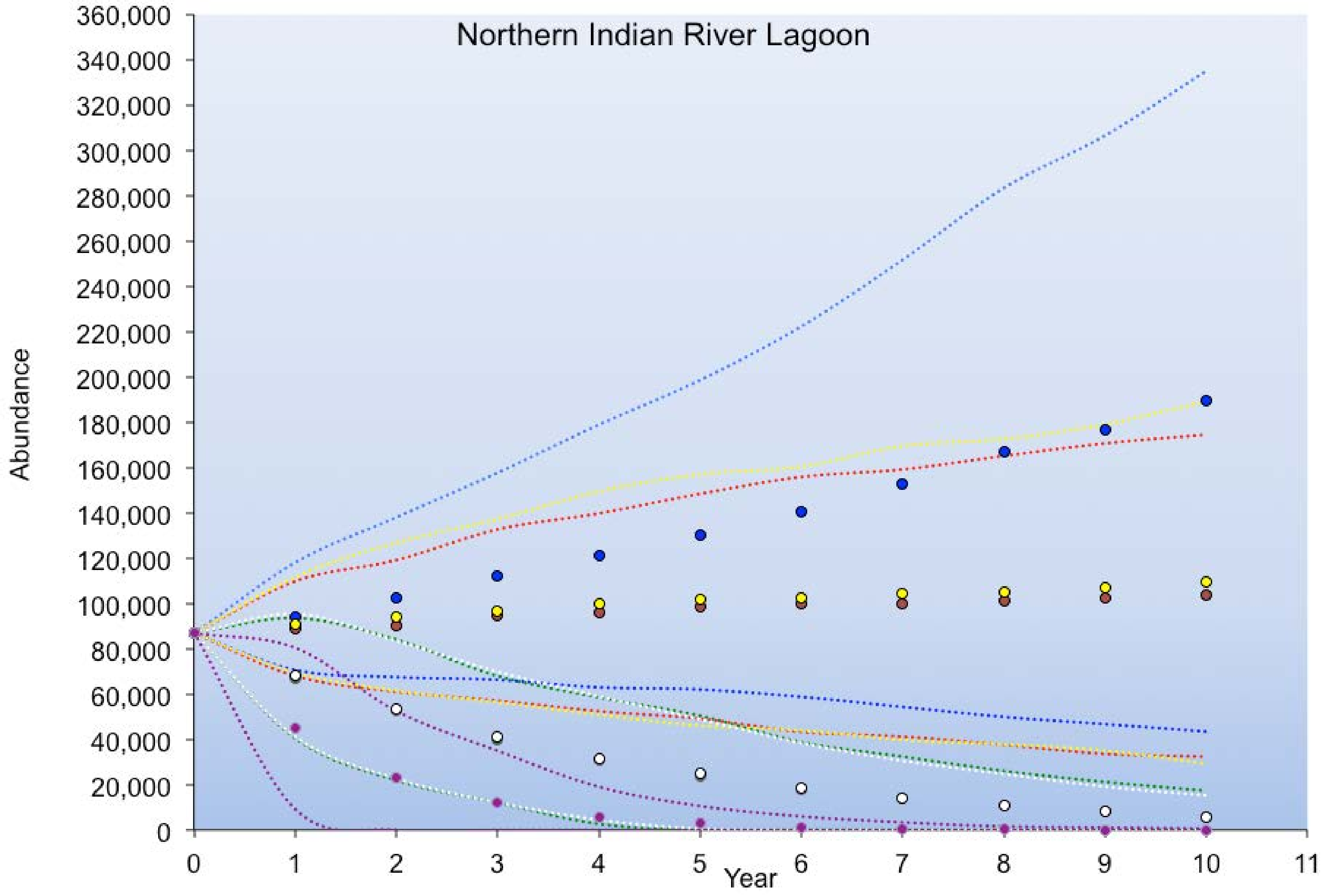

Figure 13. Projections of dwarf seahorse abundance at time t under scenarios exploring optimistic and pessimistic simulations (see Table 3 for details) for Northern Indian River Lagoon, FL. Blue circles = scenarios 25. Red circles = scenarios 26. Yellow circles = scenarios 27. Green circles = scenarios 28. White circles = scenarios 29. Purple circles = scenarios 30. Dotted lines represent ±1 standard deviation of the population mean at time t with the colors corresponding to the mean abundance symbols.