

1 **Quantifying seagrass light requirements using an algorithm to**
2 **spatially resolve depth of colonization**

3 **Marcus W. Beck¹, James D. Hagy III²**

¹ *ORISE Research Participation Program*

USEPA National Health and Environmental Effects Research Laboratory

Gulf Ecology Division, 1 Sabine Island Drive, Gulf Breeze, FL 32561

Phone: 850-934-2480, Fax: 850-934-2401, Email: beck.marcus@epa.gov

² *USEPA National Health and Environmental Effects Research Laboratory*

Gulf Ecology Division, 1 Sabine Island Drive, Gulf Breeze, FL 32561

Phone: 850-934-2455, Fax: 850-934-2401, Email: hagy.jim@epa.gov

1 Introduction

Seagrasses are ecologically valuable components of aquatic systems that serve a structural and functional role in shaping aquatic habitat. These ‘ecosystem engineers’ often govern multiple characteristics of aquatic systems through direct and indirect interactions with additional components (Jones et al. 1994, Koch 2001). For example, seagrass beds create desirable habitat for juvenile fish and invertebrates by reducing wave action and stabilizing sediment (Williams and Heck 2001, Hughes et al. 2009). Seagrasses also respond to changes in water clarity through direct physiological linkages with light availability. Seagrass communities in highly productive aquatic systems may be light-limited as increased nutrient loading may contribute to reductions in water clarity through increased algal concentration (Duarte 1995). Empirical relationships between nutrient loading, water clarity, light requirements, and the maximum depth of seagrass colonization have been identified (Duarte 1991, Kenworthy and Fonseca 1996, Choice et al. 2014) and are often used to characterize light regimes sufficient to maintain habitat through increased seagrass coverage (Steward et al. 2005). Seagrass depth limits have also been used to establish quantitative criteria for nutrient load targets for the maintenance of water quality (Janicki and Wade 1996). Seagrasses are integrative of system-wide conditions over time in relation to changes in nutrient regimes (Duarte 1995) and are often preferred biological endpoints to describe ecosystem response to perturbations relative to more variable taxa (e.g., phytoplankton). Quantifying the relationship of seagrasses with water clarity is a viable means of understanding ecological characteristics of aquatic systems with potential insights into resilience and stability of system response to disturbance (Greve and Krause-Jensen 2005).

A variety of techniques have been developed for estimating seagrass depth limits as a basis for understanding water quality dynamics and developing a more robust description of aquatic habitat. Such efforts have been useful for site-specific approaches where the analysis needs are driven by a particular management or research question (e.g., Iverson and Bittaker 1986, Hale et al. 2004). However, a lack of standardization among methods has prevented broad-scale comparisons between regions and has even contributed to discrepancies between measures of depth limits based on the chosen technique. For example, seagrass depth limits based on in situ techniques can vary with the sampling device (Spears et al. 2009). Seagrass depth limits

33 can also be estimated from geospatial data that describe aerial coverage and bathymetric depth
34 distribution. Despite the availability of such data, flexible techniques for estimating seagrass
35 depth of colonization have not been extensively developed nor have standardized techniques been
36 implemented across broad areas. Site-specific approaches typically involve the quantification of
37 depth limits within a predefined management unit as a relevant spatial context. For example,
38 Steward et al. (2005) describe use of a segmentation scheme for the Indian River Lagoon on the
39 Atlantic coast of Florida to assign seagrass depth limits to 19 distinct geospatial units. Although
40 useful within a limited scope, substantial variation in growth patterns and water quality
41 characteristics at different spatial scales may prevent more detailed analyses, thus leading to
42 limited descriptions of aquatic habitat. Methods for estimating seagrass depth limits should be
43 reproducible for broad-scale comparisons, while also maintaining flexibility of estimates
44 depending on research or management objectives. Such techniques have the potential to facilitate
45 comparisons between regions given the spatial coverage and annual availability of many
46 geospatial data sources.

47 A useful application comparing depth limit measures and water clarity is the estimation of
48 light requirements to evaluate ecologically relevant characteristics of seagrass communities.
49 Although growth of submersed aquatic plants is generally most limited by light availability
50 (Barko et al. 1982, Hall et al. 1990, Dennison et al. 1993), substantial variation for a given level of
51 light may be observed in the maximum depth of growth based on differences in light requirements
52 (Dennison et al. 1993, Choice et al. 2014). In general, seagrasses with low light requirements are
53 expected to grow deeper than seagrasses with high requirements as related to species or regional
54 differences in community attributes. Significant variation in light requirements in seagrasses
55 along the Gulf Coast of peninsular Florida were attributed to morphological and physiological
56 differences between species and adaptations to regional light regimes (Choice et al. 2014).
57 Minimum light requirements for seagrasses are on average 11% of surface irradiance (Duarte
58 1991), although values may range from less than 5% to greater than 30% at depth (Dennison et al.
59 1993). High light requirements estimated from maximum depth of colonization and water clarity
60 may suggest seagrass growth is limited by additional factors, such as high biomass of epiphytic
61 algal growth that reduces light availability on the leaf surface (Kemp et al. 2004). Spatial
62 heterogeneity in light requirements is, therefore, a useful diagnostic tool for evaluating potential

63 factors that limit seagrass growth.

64 A potentially limiting factor for estimating seagrass light requirements is the availability
65 of water clarity data that are evenly distributed through space in time, in addition to accurate
66 measures of depth of colonization. Secchi observations are routine measurements that can provide
67 consistent measures of water clarity ([USEPA, 2006](#)), although the distribution of available data
68 may limit the certainty within which light requirements can be estimated. Secchi data can be
69 biased by location such that monitoring programs may have unbalanced coverage towards aquatic
70 resources with greater perceived importance relative to those that may have more ecological
71 significance ([Wagner et al. 2008, Lottig et al. 2014](#)). Moreover, infrequent field measurements that
72 are limited to discrete time periods are often more descriptive of short-term variability rather than
73 long-term trends in water clarity ([Elsdon and Connell 2009](#)). Seagrasses growth patterns are
74 integrative of seasonal and inter-annual patterns in water clarity, among other factors, such that
75 estimates of light requirements may be limited if water clarity measurements inadequately
76 describe temporal variation. Remote sensing products can provide a reasonable estimate of water
77 clarity and could be used to develop a more spatially and temporally coherent description of
78 relevant ecosystem characteristics. Although algorithms have been developed for coastal waters
79 that relate surface reflectance to *in situ* data ([Woodruff et al. 1999, Chen et al. 2007](#)), this
80 information has rarely been used to develop a description of seagrass light requirements at a
81 spatial resolution consistent with most remote sensing products.

82 Quantitative and flexible methods for estimating seagrass depth limits and light
83 requirements have the potential to greatly improve descriptions of aquatic habitat, thus enabling
84 potentially novel insights into ecological characteristics that limit aquatic systems. This article
85 describes a method for estimating seagrass depth of colonization using geospatial datasets to
86 create a spatially-resolved and flexible measure. In particular, an empirical algorithm is described
87 that estimates seagrass depth limits from aerial coverage maps and bathymetric data using an *a*
88 *priori* defined area of influence. These estimates are combined with measures of water clarity to
89 provide a spatial characterization of light requirements to better understand factors that limit
90 seagrass growth. The specific objectives are to 1) describe the method for estimating seagrass
91 depth limits within a relevant spatial context, 2) apply the technique to four distinct regions of
92 Florida to illustrate improved clarity of description for seagrass growth patterns, and 3) develop a

93 spatial description of depth limits, water clarity, and light requirements for the case studies.
94 Overall, these methods are expected to inform the description of seagrass growth patterns to
95 develop a more ecologically relevant characterization of aquatic habitat. The method is applied to
96 data from Florida although the technique is easily transferable to other regions with comparable
97 data.

98 **2 Methods**

99 Estimates of seagrass depth of colonization (Z_c) that are derived from relatively broad
100 spatial aggregations, such as predefined management areas, may not fully describe relevant
101 variation depending on the question of interest. Fig. 1a shows variation in seagrass distribution
102 for a management segment (thick polygon) in the Big Bend region of Florida. The maximum
103 depth colonization, shown as a red countour line, is based on a segment-wide average of all
104 seagrasses within the polygon. Although such an estimate is not necessarily inaccurate,
105 substantial variation in seagrass growth patterns at smaller spatial scales is not adequately
106 described. In particular, Z_c is greatly over-estimated at the outflow of the Steinhatchee River
107 (northeast portion of the segment) where high concentrations of dissolved organic matter reduce
108 water clarity and naturally limit seagrass growth (personal communication, Nijole Wellendorf,
109 Florida Department of Environmental Protection). This example suggests that it may be useful to
110 have improved spatial resolution in estimates of Z_c , particularly when site-specific characteristics
111 may require a more detailed description of seagrass growth patterns. The following is a summary
112 of data sources, methods and rationale for developing a flexible algorithm that improves spatial
113 resolution in seagrass Z_c estimates. Data and methods described in [Hagy, In review](#) are used as a
114 foundation for developing the approach.

115 **2.1 Data sources**

116 **2.1.1 Study sites**

117 Four locations in Florida were chosen for the analysis: the Big Bend region (northeast
118 Gulf of Mexico), Choctawhatchee Bay (panhandle), Tampa Bay (central Gulf Coast), and Indian
119 River Lagoon (east coast) (Table 1 and Fig. 2). These locations represent different geographic
120 regions in the state, in addition to having available data and observed gradients in water clarity
121 that contribute to heterogeneity in seagrass growth patterns. Coastal regions and estuaries in

122 Florida are partitioned as distinct spatial units based on a segmentation scheme developed by US
123 Environmental Protection Agency (EPA) for the development of numeric nutrient criteria.
124 Site-specific estimates of seagrass depth colonization and light requirements are the primary focus
125 of the analysis, with emphasis on improved clarity of description with changes in spatial context.
126 As such, estimates that use management segments as relevant spatial units are used as a basis of
127 comparison to evaluate variation in growth patterns at difference scales. The segments included
128 the big bend region (820), Old Tampa Bay (902), and Indian River Lagoon (1502) (Fig. 2).

129 **2.1.2 Seagrass coverage and bathymetry**

130 Spatial data describing seagrass aerial coverage combined with co-located bathymetric
131 depth information were used to estimate Z_c . These geospatial data products are publically
132 available in coastal regions of Florida through the US Geological Survey, Florida Department of
133 Environmental Protection, Florida Fish and Wildlife Conservation Commission, and watershed
134 management districts. Seagrass coverage maps were obtained for recent years in each of the study
135 sites described above (Table 1). Coverage maps were produced using photo-interpretations of
136 aerial images to categorize seagrass as absent, discontinuous (patchy), or continuous. For this
137 analysis, we considered seagrass as only present (continuous and patchy) or absent since
138 differences between continuous and patchy coverage were often inconsistent between data
139 sources.

140 Bathymetric depth layers for each location were obtained from the National Oceanic and
141 Atmospheric Administration's (NOAA) National Geophysical Data Center
142 (<http://www.ngdc.noaa.gov/>) as either Digital Elevation Models (DEMs) or raw sounding data
143 from hydroacoustic surveys. Tampa Bay data provided by the Tampa Bay National Estuary
144 Program are described in Tyler et al. (2007). Bathymetric data for the Indian River Lagoon were
145 obtained from the St. John's Water Management District (Coastal Planning and Engineering
146 1997). NOAA products were referenced to mean lower low water, whereas Tampa Bay data were
147 referenced to the North American Vertical Datum of 1988 (NAVD88) and the Indian River
148 Lagoon data were referenced to mean sea level. Depth layers were combined with seagrass
149 coverage layers using standard union techniques for raster and vector layers in ArcMap 10.1
150 (Environmental Systems Research Institute 2012). To reduce computation time, depth layers were
151 first masked using a 1 km buffer of the seagrass coverage layer. Raster bathymetric layers were

{acro:EPA}

{sec:data}

{acro:DEM}

{acro:NAV}

152 converted to vector point layers to combine with seagrass coverage maps, described below. All
153 spatial data were referenced to the North American Datum of 1983 as geographic coordinates.
154 Depth values in each seagrass layer were further adjusted from the relevant vertical reference
155 datum to local mean sea level (MSL) using the NOAA VDatum tool (<http://vdatum.noaa.gov>).
156 {acro:MSL}

156 **2.1.3 Water clarity and light attenuation**

157 Seagrass light requirements can be estimated by evaluating spatial relationships between
158 depth of colonization and water clarity. These relationships were explored using Z_c estimates for
159 the whole of Tampa Bay and the Indian River Lagoon based on large gradients in water clarity
160 along longitudinal axes in each bay (cites). Satellite images were used to create a gridded map of
161 water clarity based on a previously-developed algorithm to derive light attenuation (or light
162 extinction coefficient K_Z) from surface reflectance (Chen et al. 2007). This approach was
163 preferred for Tampa Bay given the annual availability and the extent of coverage of remote
164 sensing data. Daily MODIS (Aqua level-2) data from January 2003 to December 2010 that
165 covered the spatial extent of Tampa Bay were downloaded from the NASA website
166 (<http://oceancolor.gsfc.nasa.gov>). These images were reprocessed using the SeaWiFS Data
167 Analysis System software (SeaDAS, Version 7.0). We used the clarity algorithm proposed by
168 Chen et al. (2007) to derive monthly mean and annual mean light attenuation coefficients for
169 Tampa Bay. Secchi data (meters, Z_{secchi}) were also obtained from update 40 of the Impaired
170 Waters Rule (IWR) database for all of the Indian River Lagoon (2009 coverage). Satellite
171 estimates of water clarity were unobtainable in the Indian River Lagoon because of significant
172 light scattering from bottom reflectance and limited resolution for extended narrow segments
173 along the north-south axis. Secchi data within the previous ten years of the seagrass coverage data
174 were evaluated to capture water quality trends from the most recent decade (i.e., 1999–2009 for
175 the Indian River Lagoon). Stations with less than five observations and observations that were
176 flagged indicating that the value was lower than the maximum depth of the observation point were
177 removed. Secchi data were also compared with bathymetric data to verify unflagged values were
178 not missed by initial screening.

179 **2.2 Flexible estimation of seagrass depth of colonization for finite areas**

180 The general approach to estimating seagrass depth of colonization uses combined seagrass
181 coverage maps and bathymetric depth data described above. The combined layer used for analysis

182 was a point shapefile with attributes describing location (latitude, longitude, segment), depth (m),
183 and seagrass (present, absent). Seagrass Z_c values are estimated from these data by quantifying
184 the proportion of points with seagrass at each observed depth. Three unique measures describing
185 seagrass depth limits obtained from these data are minimum ($Z_{c,min}$), median ($Z_{c,med}$), and
186 maximum ($Z_{c,max}$) depth of colonization. Operationally, these terms describe characteristics of
187 the seagrass coverage map with quantifiable significance. $Z_{c,max}$ is defined as the deepest depth
188 at which a significant coverage of mappable seagrasses occurred independent of outliers, whereas
189 $Z_{c,med}$ is the median depth occurring at the deep water edge. $Z_{c,min}$ is the depth at which seagrass
190 coverage begins to decline with increasing depth and may not be statistically distinguishable from
191 zero depth, particularly in turbid waters. Specific methods for estimating each Z_c value using
192 spatially-resolved information are described below.

193 The spatially-resolved approach for estimating Z_c begins by choosing an explicit location
194 in cartesian coordinates within the general boundaries of the available data. Seagrass depth data
195 (i.e., merged bathymetric and seagrass coverage data) that are located within a set radius from the
196 chosen location are selected for estimating seagrass Z_c values (Fig. 1). The estimate for each
197 location is quantified from a plot of the proportion of sampled points that contain seagrass at
198 decreasing 0.1 meter depth bins from the surface to the maximum observed depth in the sample
199 (Fig. 3a). Although the chosen radius for selecting depth points is problem-specific, the minimum
200 radius should be chosen to sample a sufficient number of points for estimating Z_c . In general, an
201 appropriate radius will produce a plot that indicates a decrease in the proportion of points that are
202 occupied by seagrass with increasing depth. If more than one location is used to estimate Z_c ,
203 appropriate radii for each point would have minimal overlap with the seagrass depth data sampled
204 by neighboring points.

205 A curve is fit to the sampled depth points using non-linear regression to characterize the
206 reduction in seagrass as a function of depth (Fig. 3b). Specifically, a decreasing logistic growth
207 curve is used with the assumption that seagrass decline with increasing depth is monotonic and
208 asymptotic at the minimum and maximum depths of colonization. The curve is fit by minimizing
209 the residual sums-of-squares with the Gauss-Newton algorithm (Bates and Chambers 1992) with
210 starting parameters estimated from the observed data that are initial approximations of the curve

211 characteristics. The model has the following form:

$$Proportion = \frac{\alpha}{1 + e^{(\beta - Z)/\gamma}} \quad (1) \quad \{eqn:prop\}$$

212 where the proportion of points occupied by seagrass at each depth, Z , is defined by a logistic
213 curve with an asymptote α , a midpoint inflection β , and a scale parameter γ . Finally, a simple
214 linear curve is fit through the inflection point (β) of the logistic curve to estimate the three
215 measures of depth of colonization (Fig. 3c). The inflection point is considered the depth at which
216 seagrass are decreasing at a maximum rate and is used as the slope of the linear curve. The
217 maximum depth of seagrass colonization, $Z_{c,max}$, is the x-axis intercept of the linear curve. The
218 minimum depth of seagrass growth, $Z_{c,min}$, is the location where the linear curve intercepts the
219 upper asymptote of the logistic growth curve. The median depth of seagrass colonization, $Z_{c,med}$,
220 is the depth halfway between $Z_{c,min}$ and $Z_{c,max}$. $Z_{c,med}$ is typically the inflection point of the
221 logistic growth curve.

222 Estimates for each of the three Z_c measures are obtained only if specific criteria are met.
223 These criteria were implemented as a safety measure that ensures a sufficient amount and
224 appropriate quality of data were sampled within the chosen radius. First, estimates were provided
225 only if a sufficient number of seagrass depth points were present in the sampled data to estimate a
226 logistic growth curve. This criteria applies to the sample size as well as the number of points with
227 seagrass in the sample. Second, estimates were provided only if an inflection point was present on
228 the logistic curve within the range of the sampled depth data. This criteria applied under two
229 scenarios where the curve was estimated but a trend was not adequately described by the sampled
230 data. That is, estimates were unavailable if the logistic curve described only the initial decrease
231 in points occupied as a function of depth but the observed points do not occur at depths deeper
232 than the predicted inflection point. The opposite scenario occurred when a curve was estimated
233 but only the deeper locations beyond the inflection point were present in the sample. Third, the
234 estimate for $Z_{c,min}$ was set to zero depth if the linear curve through the inflection point
235 intercepted the asymptote at x-axis values less than zero. The estimate for $Z_{c,med}$ was also shifted
236 to the depth value halfway between $Z_{c,min}$ and $Z_{c,max}$ if $Z_{c,min}$ was fixed at zero. Finally,
237 estimates were considered invalid if the 95% confidence interval for $Z_{c,max}$ included zero.

238 Methods used to determine confidence bounds on Z_c estimates are described below.

239 2.3 Estimating uncertainty in depth of colonization estimates

240 Confidence intervals for the Z_c values were estimated using a Monte Carlo simulation
241 approach that considered the variance and covariance between the model parameters (Hilborn and
242 Mangel 1997). For simplicity, we assume that the variability associated with parameter estimates
243 is the dominant source of uncertainty. A 95% confidence interval for each Z_c estimate was
244 constructed by repeated sampling of a multivariate normal distribution followed by prediction of
245 the proportion of points occupied by seagrass as in eq. (1). The sampling distribution assumes:

$$x \sim N(\mu, \Sigma) \quad (2)$$

246 where x is a predictor variable used in eq. (1) (depth) that follows a multivariate normal
247 distribution with mean μ , and variance-covariance matrix Σ . The mean values are set at the depth
248 value corresponding to the inflection point on the logistic curve and the predicted model
249 parameters (i.e., α , β , and γ), whereas Σ is the variance-covariance matrix of the model
250 parameters. A large number of samples ($n = 10000$) were drawn from the distribution to
251 characterize the uncertainty of the depth value at the inflection point. The 2.5th and 97.5th quantile
252 values of the sample were considered bounds on the 95% confidence interval.

253 The uncertainty associated with the Z_c estimates was based on the upper and lower limits
254 of the estimated inflection point on the logistic growth curve. This approach was used because
255 uncertainty in the inflection point is directly related to uncertainty in each of the Z_c estimates that
256 are based on the linear curve fit through the inflection point. Specifically, linear curves were fit
257 through the upper and lower estimates of the depth value at the inflection point to identify upper
258 and lower limits for the estimates of $Z_{c,min}$, $Z_{c,med}$, and $Z_{c,max}$. These values were compared
259 with the initial estimates from the linear curve that was fit through the inflection point on the
260 predicted logistic curve (i.e., Fig. 3c). This approach provided an indication of uncertainty for
261 individual estimates for the chosen radius. Uncertainty estimates were obtained for each Z_c
262 estimate for the grids in each segment.

263 The algorithm for estimating Z_c was implemented custom-made and pre-existing
264 functions in program R. Nonlinear least squares models were based on the `nls` and `SSlogis`

265 functions that used a self-starting logistic growth model (Bates and Chambers 1992, R
266 Development Core Team 2014). Multivariate normal distributions used to evaluate uncertainty
267 were simulated using functions in the MASS package (Venables and Ripley 2002). Geospatial
268 data were imported and processed using functions in the rgeos and sp packages (Bivand et al.
269 2008, Bivand and Rundel 2014).

270 **2.4 Evaluation of spatial heterogeneity of seagrass depth limits**

271 Spatially-resolved estimates for seagrass Z_c were obtained for each of the four coastal
272 segments described above. Segment-wide estimates obtained using all data were used as a basis
273 of comparison such that departures from these values at smaller scales were evidence of spatial
274 heterogeneity in seagrass growth patterns and improved clarity of description in depth estimates.
275 A sampling grid of locations for estimating each of the three depth values in Fig. 3 was created
276 for each segment. The grid was masked by the segment boundaries, whereas seagrass depth
277 points used to estimate Z_c extended beyond the segment boundaries to allow sampling by grid
278 points that occurred near the edge of the segment. Initial spacing between sample points was
279 chosen arbitrarily as 0.01 decimal degrees, which is approximately 1 km at 30 degrees N latitude.
280 The sampling radius around each sampling location in the grid was also chosen as 0.02 decimal
281 degrees to allow for complete coverage of seagrass within the segment while also minimizing
282 redundancy of information described by each location. In other words, radii were chosen such
283 that the seagrass depth points sampled by each grid location were only partially overlapped by
284 those sampled by neighboring points, while also ensuring an adequate number of locations were
285 sampled that included seagrass.

286 **2.5 Developing a spatially coherent relationship of water clarity with depth 287 of colonization**

288 The relationship between the quantified seagrass depth limits and secchi measurements
289 were explored by estimating light requirements from standard attenuation equations. The
290 traditional Lambert-Beer equation describes the exponential decrease of light availability with
291 depth:

$$I_z = I_O \cdot \exp(-K_d \cdot Z) \quad (3) \quad \{\text{eqn: lambert}\}$$

such that the irradiance of incident light at depth Z (I_Z) can be estimated from the irradiance at the surface (I_O) and a light extinction coefficient (K_d). Light requirements of seagrass at a specific location can be estimated by rearranging eq. (3):

$$\% \text{ light} = \exp(-K_d \cdot Z_{c, \max}) \quad (4) \quad \{\text{eqn:perc}\}$$

where the percent light requirements of seagrass at $Z_{c, \max}$ are empirically related to light extinction. A conversion factor is often used to estimate the light extinction coefficient from secchi depth Z_{secchi} , such that $c = K_d \cdot Z_{secchi}$, where c has been estimated as 1.7 (Poole and Atkins 1929, Idso and Gilbert 1974). Thus, K_d can be replaced with the conversion factor and Z_{secchi} :

$$\% \text{ light} = \exp\left(-\left(\frac{1.7}{Z_{secchi}}\right) \cdot Z_{c, \max}\right) \quad (5) \quad \{\text{eqn:cperc}\}$$

Variation in seagrass light requirements by location can be considered biologically meaningful.

An evenly-spaced grid of sampling points was created for the spatial extent of Tampa Bay to estimate light requirements for seagrasses. Grid spacing was set at 0.01 decimal degrees as before. These points were used to sample the raster grid of satellite-derived water clarity and the seagrass depth points to estimate $Z_{c, \max}$. Similarly, the geographic coordinates for each available secchi measurement in the Indian River Lagoon were used as locations for estimating $Z_{c, \max}$. These estimates were compared with the averaged water clarity or secchi data for all preceding years to identify seagrass light requirements at each location (i.e., 2003–2010 for Tampa Bay and 1999–2009 for Indian River Lagoon). However, the relationship may vary depending on the specific radius around each sample point for estimating $Z_{c, \max}$. A sufficiently large radius was chosen that was an order of magnitude larger than that used for the individual segments given that $Z_{c, \max}$ estimates were to be compared for whole bays rather than within segments. The estimated maximum depth values and light requirements of each point were plotted by location to evaluate spatial variation in seagrass growth as a function of light-limitation.

314 **3 Results**

315 **3.1 Segment characteristics and seagrass depth estimates**

316 Each of the four segments varied by several key characteristics that potentially explain
317 within-segment variation of seagrass growth patterns (Table 1). Mean surface area was 191.2
318 square kilometers, with area decreasing for the Big Bend (271.4 km), Indian River Lagoon (NA
319 km), Old Tampa Bay (205.5 km), and Choctawhatchee Bay (59.4 km) segments. Seagrass
320 coverage as a percentage of total surface area varied considerably by segment. Seagrasses covered
321 a majority of the surface area for the Big Bend segment (74.8 %), whereas coverage was much
322 less for Indian River Lagoon (NA %), Old Tampa Bay (11.9 %), and Choctawhatchee Bay (5.9
323 %). Visual examination of the seagrass coverage maps for the respective year of each segment
324 suggested that seagrasses were not uniformly distributed (Fig. 2). Seagrasses in the
325 Choctawhatchee Bay segments were generally sparse with the exception of a large patch located
326 to the west of the inlet connection with the Gulf of Mexico. Seagrasses in the Big Bend segment
327 were located throughout the segment with noticeable declines near the outflow of the
328 Steinhatchee River, whereas seagrasses in Old Tampa Bay and the Indian River Lagoon segment
329 were generally confined to shallow areas near the shore. Seagrass coverage showed a partial
330 decline toward the northern ends of both Old Tampa Bay and the Indian River Lagoon segments.
331 Mean depth was less than 5 meters for each segment, excluding Choctawhatchee Bay which was
332 slightly deeper than the other segments on average (5.3 m). Maximum depths were considerably
333 deeper for Choctawhatchee Bay (11.9 m) and Old Tampa Bay (10.4 m), as compared to the Big
334 Bend (3.6 m) and Indian River Lagoon (NA m) segments. Water clarity as indicated by average
335 secchi depths was similar between the segments (1.5 m), although Choctawhatchee Bay had a
336 slightly higher average (2.1 m).

337 Estimates of seagrass Z_c using a segment-wide approach that did not consider spatially
338 explicit locations indicated that seagrasses generally did not grow deeper than three meters in any
339 of the segments (Table 2). Maximum and median depth of colonization were deepest for the Big
340 Bend segment (3.7 and 2.5 m, respectively) and shallowest for Old Tampa Bay (1.1 and 0.9 m),
341 whereas the minimum depth of colonization was deepest for Choctawhatchee Bay (1.8 m) and
342 shallowest for Old Tampa Bay (0.6 m). Averages of all grid-based estimates for each segment

were different than the segment wide estimates, which suggests potential bias associated with using a whole segment as a relevant spatial unit for estimating depth of colonization. In most cases, the averages of all grid-based estimates were less than the whole segment estimates, suggesting the latter provided an over-estimate of seagrass growth limits. For example, the average of all grid estimates for $Z_{c, max}$ in the Big Bend region suggested seagrasses grew to approximately 2.1 m, which was 1.6 m less than the whole segment estimate. This reduction is likely related to improved resolution of seagrass depth limits near the outflow of the Steinhatchee river. Although reductions were not as severe for the average grid estimates for the remaining segments, considerable within-segment variation was observed depending on grid location. For example, the deepest estimate for $Z_{c, min}$ (2 m) in the Indian River Lagoon exceeded the average of all grid locations for $Z_{c, max}$ (1.7 m). $Z_{c, min}$ also had minimum values of zero meters for the Big Bend and Old Tampa Bay segments, suggesting that seagrasses declined continuously from the surface for several locations.

Visual interpretations of seagrass depth estimates using the grid-based approach provided further information on the distribution of seagrasses in each segment (Fig. 4). Spatial heterogeneity in depth limits was particularly apparent for the Big Bend and Indian River Lagoon segments. As expected, depth estimates indicated that seagrasses grew deeper at locations far from the outflow of the Steinhatchee River in the Big Bend segment. Similarly, seagrasses were limited to shallower depths at the north end of the Indian River Lagoon segment near the Merrit Island National Wildlife Refuge. Seagrasses were estimated to grow at maximum depths up to 2.2 m on the eastern portion of the Indian River Lagoon segment. Spatial heterogeneity was less distinct for the remaining segments. Seagrasses in Old Tampa Bay grew deeper in the northeast portion of the segment and declined to shallower depths near the inflow at the northern edge. Spatial variation in the Choctawhatchee Bay segment was not apparent, although the maximum Z_c estimate was observed in the northeast portion of the segment. Z_c values were not available for all grid locations given the limitations imposed in the estimation method. Z_c could not be estimated in locations where seagrasses were sparse or absent, nor where seagrasses were present but the sampled points did not exhibit a sufficient decline with depth. The latter scenario was most common in Old Tampa Bay and Choctawhatchee Bay where seagrasses were unevenly distributed or confined to shallow areas near the shore. The former scenario was most common in

373 the Big Bend segment where seagrasses were abundant but locations near the shore were
374 inestimable given that seagrasses did not decline appreciably within the depths that were sampled.

375 Uncertainty for estimates of $Z_{c,max}$ indicated that confidence intervals were generally
376 acceptable (i.e., greater than zero), although the ability to discriminate between the three depth
377 estimates varied by segment (?? and Table 3). Mean uncertainty for all estimates in each segment
378 measured as the width of a 95% confidence interval was 0.2 m. Greater uncertainty was observed
379 for Choctawhatchee Bay (mean width of all confidence intervals was 0.5 m) and Old Tampa Bay
380 (0.4 m), compared to the Big Bend (0.1 m) and Indian River Lagoon (0.1 m) segments. The
381 largest confidence interval for each segment was 1.4 m for Old Tampa Bay, 1.6 m for
382 Choctawhatchee Bay, 0.5 m for the Big Bend, and 0.8 m for the Indian River Lagoon segments.
383 However, most confidence intervals for the remaining grid locations were much smaller than the
384 maximum in each segment. A comparison of overlapping confidence intervals for $Z_{c,min}$, $Z_{c,med}$,
385 and $Z_{c,max}$ at each grid location indicated that not every measure was unique. Specifically, only
386 11.1% of grid points in Choctawhatchee Bay and 28.2% in Old Tampa Bay had significantly
387 different estimates, whereas 82.4% of grid points in the Indian River Lagoon and 96.2% of grid
388 points in the Big Bend segments had estimates that were significantly different. By contrast, all
389 grid estimates in Choctawhatchee Bay and Indian River Lagoon had $Z_{c,max}$ estimates that were
390 significantly greater than zero, whereas all but 12.4% of grid points in Old Tampa Bay and 8% of
391 grid points in the Big Bend segment had $Z_{c,max}$ estimates significantly greater than zero.

392 3.2 Evaluation of seagrass light requirements

393 Estimates of seagrass depth limits and corresponding light requirements for all segments
394 of Tampa Bay and the Indian River Lagoon indicated substantial variation, both between and
395 within the different bays (Table 4 and Figs. 8 and 9). Seagrass Z_c estimates were obtained for 566
396 locations in Tampa Bay and 50 locations in the Indian River Lagoon where secchi observations
397 were available in the Florida IWR database. Mean secchi depth for all recorded observations was
398 2.3 m ($n = 566$) for Tampa Bay and NA m ($n = 50$) for Indian River Lagoon. Mean light
399 requirements were significantly different between the bays (two-sided t-test, $t = 19$, $df = 62.7$,
400 $p < 0.001$) with a mean requirement of 30.4% for Tampa Bay and 10.6% for Indian River
401 Lagoon. Within each bay, light requirements were significantly different between segments
402 (ANOVA, $F = 84.6$, $df = 3, 562$, $p = 0.00$ for Tampa Bay, $F = 5.2$, $df = 7, 42$, $p = 0.000$ for

403 Indian River Lagoon). However, post-hoc evaluation of all pair-wise comparisons of mean light
404 requirements indicated that significant differences were only observed between a few segments
405 within each bay. Significant differences in Tampa Bay were observed between Old Tampa Bay
406 and Hillsborough Bay (Tukey multiple comparisons, $p = 0.003$). Significant differences in the
407 Indian River Lagoon were observed between the Upper Indian River Lagoon and Banana River
408 ($p = 0.915$), the Upper Indian River Lagoon and Lower Indian River Lagoon ($p = 0.140$), and
409 Upper Indian River Lagoon and Lower St. Lucie ($p = 0.103$) segments. In general, spatial
410 variation of light requirements in Tampa Bay suggested that seagrasses were less light-limited
411 (i.e., lower percent light requirements at $Z_{c, max}$) in Hillsborough Bay and western areas of Lower
412 Tampa Bay near the Gulf of Mexico (Fig. 8). Seagrassess in the Indian River Lagoon were
413 generally less light-limited towards the south and in the Banana River segment (Fig. 9).

414 **4 Discussion**

415 **References**

- 416 Barko JW, Hardin DG, Matthews MS. 1982. Growth and morphology of submersed freshwater
417 macrophytes in relation to light and temperature. Canadian Journal of Botany, 60(6):877–887.
- 418 Bates DM, Chambers JM. 1992. Nonlinear models. In: Chambers JM, Hastie TJ, editors,
419 Statistical Models in S, pages 421–454. Wadsworth and Brooks/Cole, Pacific Grove, California.
- 420 Bivand R, Rundel C. 2014. rgeos: Interface to Geometry Engine - Open Source (GEOS). R
421 package version 0.3-8.
- 422 Bivand RS, Pebesma EJ, Gómez-Rubio V. 2008. Applied Spatial Data Analysis with R. Springer,
423 New York, New York.
- 424 Chen Z, Muller-Karger FE, Hu C. 2007. Remote sensing of water clarity in Tampa Bay. Remote
425 Sensing of Environment, 109(2):249–259.
- 426 Choice ZD, Frazer TK, Jacoby CA. 2014. Light requirements of seagrasses determined from
427 historical records of light attenuation along the Gulf coast of peninsular Florida. Marine
428 Pollution Bulletin, 81(1):94–102.
- 429 Coastal Planning and Engineering. 1997. Indian River Lagoon bathymetric survey. A final report
430 to St. John's River Water Management District. Technical Report Contract 95W142, Coastal
431 Planning and Engineering, Palatka, Florida.
- 432 Dennison WC, Orth RJ, Moore KA, Stevenson JC, Carter V, Kollar S, Bergstrom PW, Batiuk RA.
433 1993. Assessing water quality with submersed aquatic vegetation. BioScience, 43(2):86–94.
- 434 Duarte CM. 1991. Seagrass depth limits. Aquatic Botany, 40(4):363–377.
- 435 Duarte CM. 1995. Submerged aquatic vegetation in relation to different nutrient regimes.
436 Ophelia, 41:87–112.
- 437 Elsdon TS, Connell SD. 2009. Spatial and temporal monitoring of coastal water quality: refining
438 the way we consider, gather, and interpret patterns. Aquatic Biology, 5(2):157–166.
- 439 Environmental Systems Research Institute. 2012. ArcGIS v10.1. ESRI, Redlands, California.
- 440 Greve T, Krause-Jensen D. 2005. Stability of eelgrass (*Zostera marina L.*) depth limits:
441 influence of habitat type. Marine Biology, 147(3):803–812.
- 442 Hagy JD. In review. Seagrass depth of colonization in Florida estuaries.
- 443 Hale JA, Frazer TK, Tomasko DA, Hall MO. 2004. Changes in the distribution of seagrass species
444 along Florida's central gulf coast: Iverson and Bittaker revisited. Estuaries, 27(1):36–43.
- 445 Hall MO, Durako MJ, Fourqurean JW, Zieman JC. 1990. Decadal changes in seagrass
446 distribution and abundance in Florida Bay. Estuaries, 22(2B):445–459.

- 447 Hilborn R, Mangel M. 1997. The Ecological Detective: Confronting Models with Data.
448 Princeton University Press, Princeton, New Jersey.
- 449 Hughes AR, Williams SL, Duarte CM, Heck KL, Waycott M. 2009. Associations of concern:
450 declining seagrasses and threatened dependent species. *Frontiers in Ecology and the*
451 *Environment*, 7(5):242–246.
- 452 Idso SB, Gilbert RG. 1974. On the universality of the Poole and Atkins secchi disk-light
453 extinction equation. *Journal of Applied Ecology*, 11(1):399–401.
- 454 Iverson RL, Bittaker HF. 1986. Seagrass distribution and abundance in eastern Gulf of Mexico
455 coastal waters. *Estuarine, Coastal and Shelf Science*, 22(5):577–602.
- 456 Janicki A, Wade D. 1996. Estimating critical external nitrogen loads for the Tampa Bay estuary:
457 An empirically based approach to setting management targets. Technical Report 06-96, Tampa
458 Bay National Estuary Program, St. Petersburg, Florida.
- 459 Jones CG, Lawton JH, Shachak M. 1994. Organisms as ecosystem engineers. *OIKOS*,
460 69(3):373–386.
- 461 Kemp WC, Batiuk R, Bartleson R, Bergstrom P, Carter V, Gallegos CL, Hunley W, Karrh L, Koch
462 EW, Landwehr JM, Moore KA, Murray L, Naylor M, Rybicki NB, Stevenson JC, Wilcox DJ.
463 2004. Habitat requirements for submerged aquatic vegetation in Chesapeake Bay: Water
464 quality, light regime, and physical-chemical factors. *Estuaries*, 27(3):363–377.
- 465 Kenworthy WJ, Fonseca MS. 1996. Light requirements of seagrasses *Halodule wrightii* and
466 *Syringodium filiforme* derived from the relationship between diffuse light attenuation and
467 maximum depth distribution. *Estuaries*, 19(3):740–750.
- 468 Koch EW. 2001. Beyond light: Physical, geological, and geochemical parameters as possible
469 submersed aquatic vegetation habitat requirements. *Estuaries*, 24(1):1–17.
- 470 Lottig NR, Wagner T, Henry EN, Cheruvellil KS, Webster KE, Downing JA, Stow CA. 2014.
471 Long-term citizen-collected data reveal geographical patterns and temporal trends in water
472 clarity. *PLoS ONE*, 9(4):e95769.
- 473 Poole HH, Atkins WRG. 1929. Photo-electric measurements of submarine illumination
474 throughout the year. *Journal of the Marine Biological Association of the United Kingdom*,
475 16:297–324.
- 476 R Development Core Team. 2014. R: A language and environment for statistical computing,
477 v3.1.2. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org>.
- 478 Spears BM, Gunn IDM, Carvalho L, Winfield IJ, Dudley B, Murphy K, May L. 2009. An
479 evaluation of methods for sampling macrophyte maximum colonisation depth in Loch Leven,
480 Scotland. *Aquatic Botany*, 91(2):75–81.
- 481 Steward JS, Virnstein RW, Morris LJ, Lowe EF. 2005. Setting seagrass depth, coverage, and light
482 targets for the Indian River Lagoon system, Florida. *Estuaries*, 28(6):923–935.

- 483 Tyler D, Zawada DG, Nayegandhi A, Brock JC, Crane MP, Yates KK, Smith KEL. 2007.
- 484 Topobathymetric data for Tampa Bay, Florida. Technical Report Open-File Report 2007-1051
- 485 (revised), US Geological Survey, US Department of the Interior, St. Petersburg, Florida.
- 486 USEPA (US Environmental Protection Agency). 2006. Volunteer estuary monitoring: A methods
- 487 manual, second edition. Technical Report EPA-842-B-06-003, Washington, DC.
- 488 Venables WN, Ripley BD. 2002. Modern Applied Statistics with S. Springer, New York, New
- 489 York, fourth edition.
- 490 Wagner T, Soranno PA, Cheruvellil KS, Renwick WH, Webster KE, Vaux P, Abbott RJ. 2008.
- 491 Quantifying sample biases of inland lake sampling programs in relation to lake surface area and
- 492 land use/cover. *Environmental Monitoring and Assessment*, 141(1-3):131–147.
- 493 Williams SL, Heck KL. 2001. Seagrass community ecology. In: Bertness MD, Gaines SD, Hay
- 494 ME, editors, *Marine Community Ecology*. Sinauer Associates, Sunderland, Massachusetts.
- 495 Woodruff DL, Stumpf RP, Scope JA, Paerl HW. 1999. Remote estimation of water clarity in
- 496 optically complex estuarine waters. *Remote Sensing of Environment*, 68(1):41–52.

Table 1: Characteristics of coastal segments used to evaluate seagrass depth of colonization estimates (see Fig. 2 for spatial distribution). Year is the date of the seagrass coverage and bathymetric data. Latitude and longitude are the geographic centers of each segment. Area and depth values are meters and square kilometers, respectively. Secchi measurements (m) were obtained from the Florida Department of Environmental Protection’s Impaired Waters Rule (IWR) database, update number 40. Secchi mean and standard errors are based on all observations within the ten years preceding each seagrass survey.^{tab:seg_summ}

	Big Bend	Choctawhatchee Bay	Old Tampa Bay	Upper Indian R. Lagoon
Year ^a	2006	2007	2010	2009
Latitude	29.61	30.43	27.94	28.61
Longitude	-83.48	-86.54	-82.62	-80.77
Surface area	271.37	59.41	205.50	228.52
Seagrass area	203.02	3.51	24.48	74.89
Depth (mean)	1.41	5.31	2.56	1.40
Depth (max)	3.60	11.90	10.40	3.70
Secchi (mean)	1.34	2.14	1.41	1.30
Secchi (se)	0.19	0.08	0.02	0.02

^a Seagrass coverage data sources, see section 2.1.2 for bathymetry data sources:

Big Bend: http://atoll.floridamarine.org/Data/metadata/SDE_Current/seagrass_bigbend_2006_poly.htm

Choctawhatchee Bay: http://atoll.floridamarine.org/data/metadata/SDE_Current/seagrass_chotawhatchee_2007_poly.htm

Tampa Bay: http://www.swfwmd.state.fl.us/data/gis/layer_library/category/swim

Indian R. Lagoon: <http://www.sjrwmd.com/gisdevelopment/docs/themes.html>

Table 2: Summary of seagrass depth estimates (m) for each segment using all grid locations in Fig. 4. Whole segment estimates were obtained from all seagrass depth data for each segment.^{tab:est_summ}

Segment ^a	Whole segment	Mean	St. Dev.	Min	Max
BB					
$Z_{c, min}$	1.25	1.39	0.77	0.00	2.68
$Z_{c, med}$	2.46	1.74	0.76	0.47	2.90
$Z_{c, max}$	3.66	2.09	0.80	0.74	3.33
CB					
$Z_{c, min}$	1.82	1.56	0.50	0.44	2.23
$Z_{c, med}$	2.16	1.93	0.37	1.26	2.49
$Z_{c, max}$	2.50	2.30	0.39	1.63	2.99
OTB					
$Z_{c, min}$	0.61	0.60	0.29	0.00	1.23
$Z_{c, med}$	0.88	0.90	0.29	0.30	1.64
$Z_{c, max}$	1.15	1.19	0.38	0.37	2.16
UIRL					
$Z_{c, min}$	1.25	1.35	0.25	0.81	2.01
$Z_{c, med}$	1.51	1.52	0.23	0.97	2.08
$Z_{c, max}$	1.77	1.69	0.23	1.06	2.22

^aBB: Big Bend, CB: Choctawhatchee Bay, OTB: Old Tampa Bay, UIRL: Upper Indian River Lagoon.

Table 3: Summary of uncertainty for seagrass depth estimates (m) for each segment using all grid locations in ???. The uncertainty values are equally applicable to each seagrass depth measure ($Z_{c, min}$, $Z_{c, med}$, $Z_{c, max}$).^{tab:sens_summ}

Segment ^a	Mean	St. Dev	Min	Max
BB	0.10	0.09	0.01	0.49
CB	0.53	0.37	0.12	1.57
OTB	0.38	0.26	0.06	1.40
UIRL	0.10	0.10	0.00	0.81

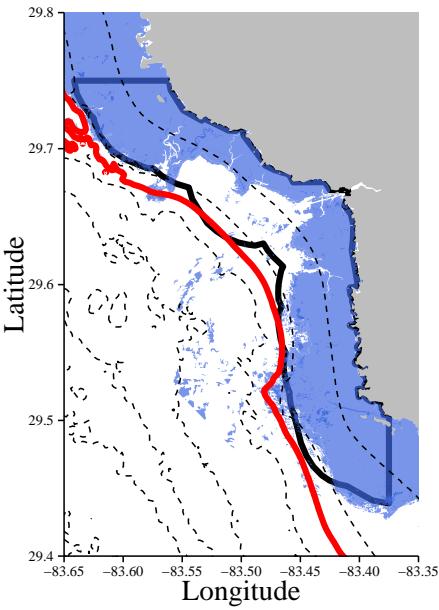
^aBB: Big Bend, CB: Choctawhatchee Bay, OTB: Old Tampa Bay, UIRL: Upper Indian River Lagoon.

Table 4: Summary of water clarity data (Z_{secchi}), depth of colonization ($Z_{c,max}$), and estimated light requirements for bay segments with available data for the Indian River Lagoon and Tampa Bay. Water clarity data were obtained from secchi observations in the Florida Impaired Waters Rule database for all available locations and dates within ten years of the seagrass survey in each bay. Values are minimum and maximum years of secchi data, sample size of secchi data ($n_{Z_{secchi}}$), mean values (m) of secchi data, sample size of seagrass depth estimates ($n_{Z_{c,max}}$) at each unique secchi location, mean $Z_{c,max}$, and estimated % light requirements for each segment. See Figs. 8 and 9 for spatial distribution of the results.^a

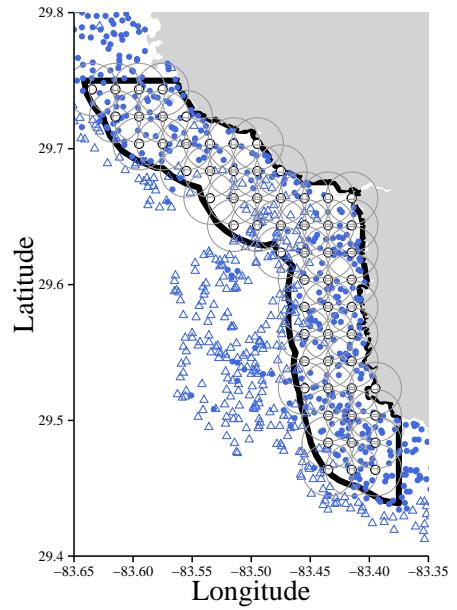
Segment ^a	Min year	Max year	$n_{Z_{secchi}}$	Z_{secchi}	$n_{Z_{c,max}}$	$Z_{c,max}$	% light
Indian River Lagoon							
BR	2000	2009	899	1.06	2	1.38	11.96
LCIRL	2000	2009	644	1.02	12	1.41	9.23
LIRL	2000	2005	111	0.93	6	1.84	4.06
LML	2000	2009	217	1.14	4	1.14	17.84
LSL	2000	2005	52	0.94	3	2.37	2.02
UCIRL	2000	2009	1148	1.14	18	1.19	10.84
UIRL	2000	2009	593	1.30	1	1.15	20.32
UML	2000	2009	258	1.03	4	1.21	19.08
Tampa Bay							
HB	2001	2003	412	1.25	48	1.26	57.23
LTB	2001	2009	807	2.47	155	2.20	48.28
MTB	2001	2009	570	2.19	212	1.76	52.89
OTB	2001	2003	671	1.44	137	1.17	60.06

^aBR: Banana R., LCIRL: Lower Central Indian R. Lagoon, LIRL: Lower Indian R. Lagoon, LML: Lower Mosquito Lagoon, LSL: Lower St. Lucie, UCIRL: Upper Central Indian R. Lagoon, UIRL: Upper Indian R. Lagoon, UML: Upper Mosquito Lagoon, HB: Hillsborough Bay, LTB: Lower Tampa Bay, MTB: Middle Tampa Bay, OTB: Old Tampa Bay.

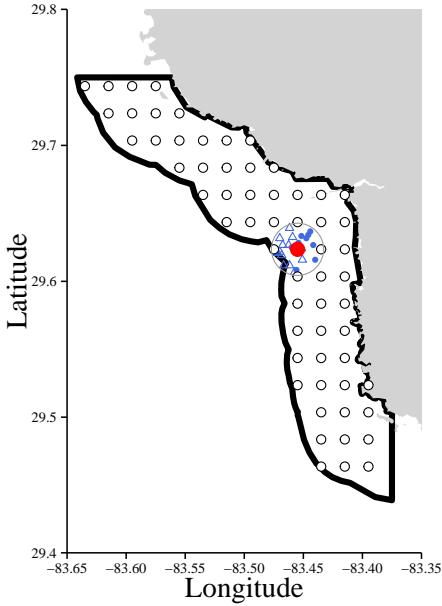
(a) Seagrass coverage and bathymetry for the segment



(b) Grid of locations and sample areas for estimates



(c) Sampled seagrass data for a test point



- Seagrass coverage
- 2 m depth contours
- Estimated depth limit for segment
- ▨ Segment polygon

- △ Seagrass absent
- Seagrass present

- Estimation grid
- Test point
- Sample area

Fig. 1: Examples of data and grid locations for estimating seagrass depth of colonization for a region of the Big Bend, Florida. Fig. 1a shows the seagrass coverage and depth contours at 2 meter intervals, including the whole segment estimate for depth of colonization. Fig. 1b shows a grid of sampling locations with sampling radii for estimating Z_c and seagrass depth points derived from bathymetry and seagrass coverage layers. Fig. 1c shows an example of sampled seagrass depth points for a test location. Estimates in Fig. 3 were obtained from the test location in Fig. 1c.

{fig:buff_}

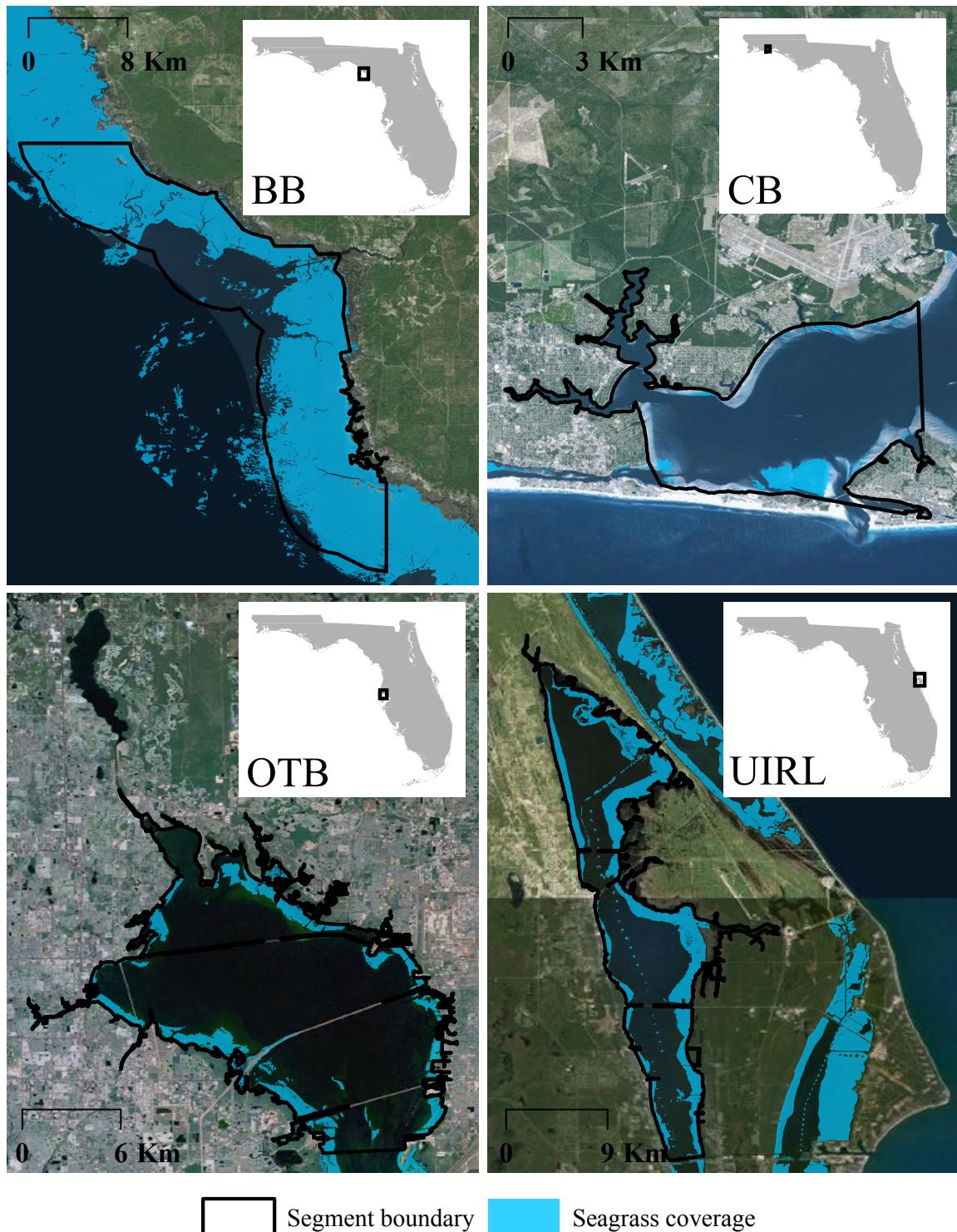
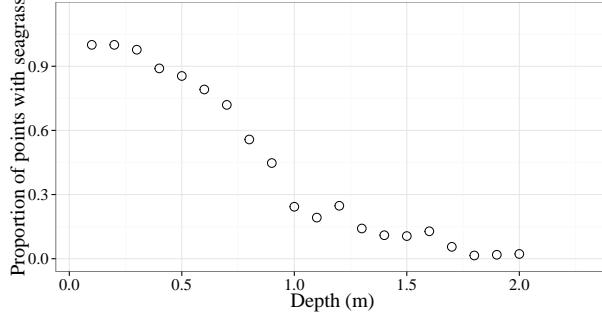


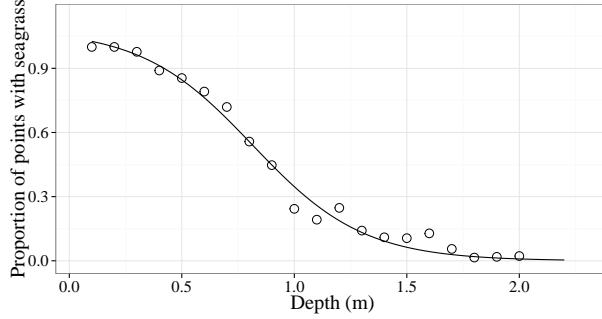
Fig. 2: Locations and seagrass coverage of estuary segments used to evaluate depth of colonization estimates. Seagrass coverage layers are from 2007 (CB: Choctawhatchee Bay), 2006 (BB: Big Bend), 2010 (OTB: Old Tampa Bay), and 2009 (UIRL: Upper Indian R. Lagoon).

{fig:seg_a}

(a) Proportion of points with seagrass by depth



(b) Logistic growth curve fit through points



(c) Depth estimates

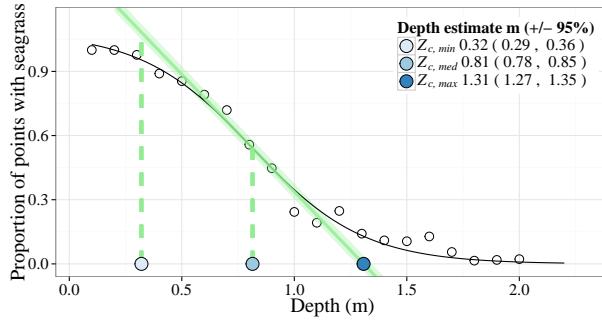


Fig. 3: Methods for estimating seagrass depth of colonization using sampled seagrass depth points around a single location. Fig. 3a is the proportion of points with seagrass by depth using depth points within the buffer of the test point in Fig. 1. Fig. 3b adds a decreasing logistic growth curve fit through the points. Fig. 3c shows three depth estimates based on a linear curve fit through the inflection point of logistic growth curve.

{fig:est_e}

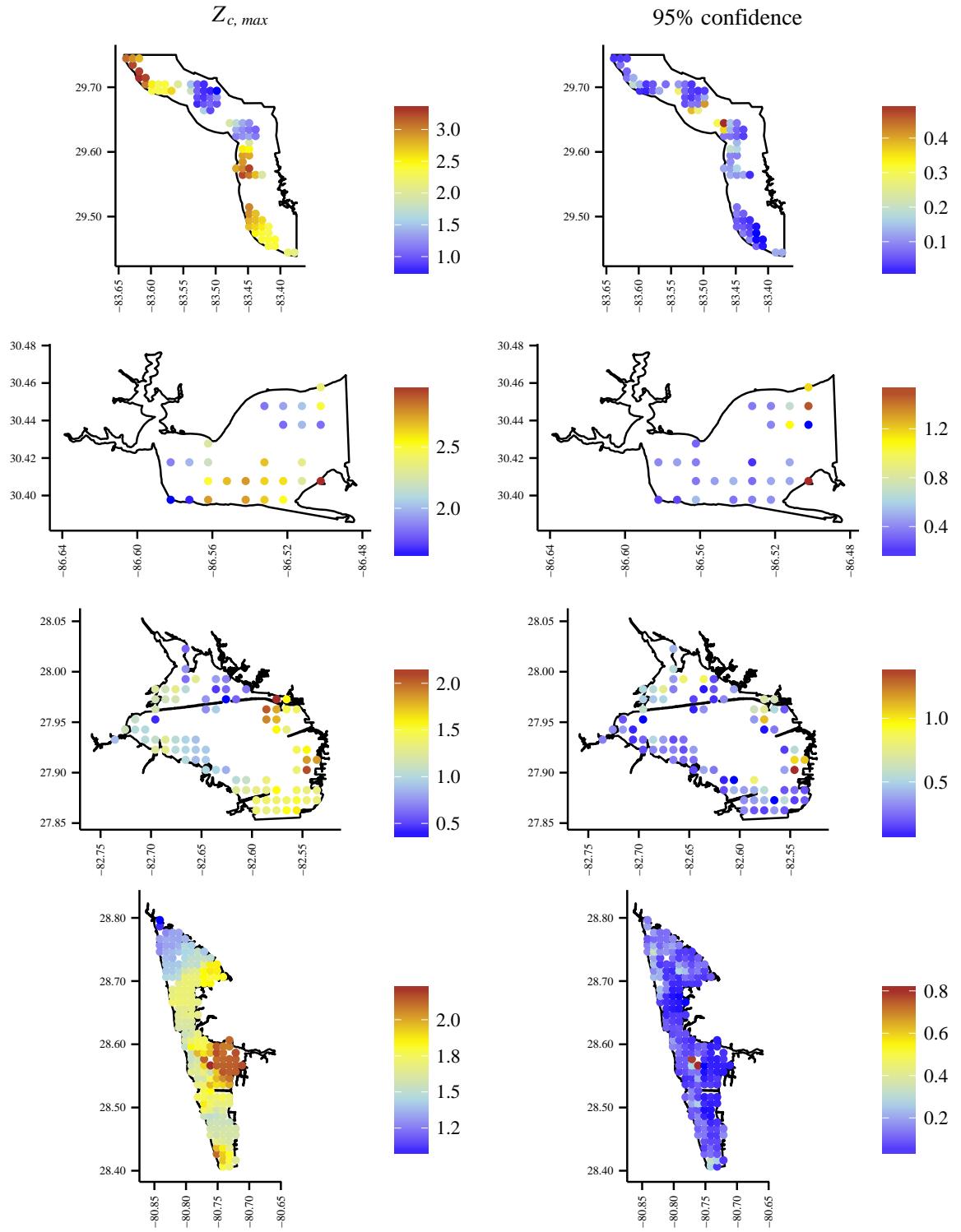


Fig. 4: Spatially-resolved estimates of seagrass depth limits (m) for four coastal segments of Florida. Maximum depth of colonization ($Z_{c, max}$) estimates are on the left and width of 95% confidence intervals are on the right. Estimates are assigned to grid locations for each segment, where grid spacing was fixed at 0.02 decimal degrees. Radii for sampling seagrass bathymetric data around each grid location were fixed at 0.06 decimal degrees. From top to bottom: Big Bend, Choctawhatchee Bay, Old Tampa Bay, Upper Indian R. Lagoon.

{fig:all_e}

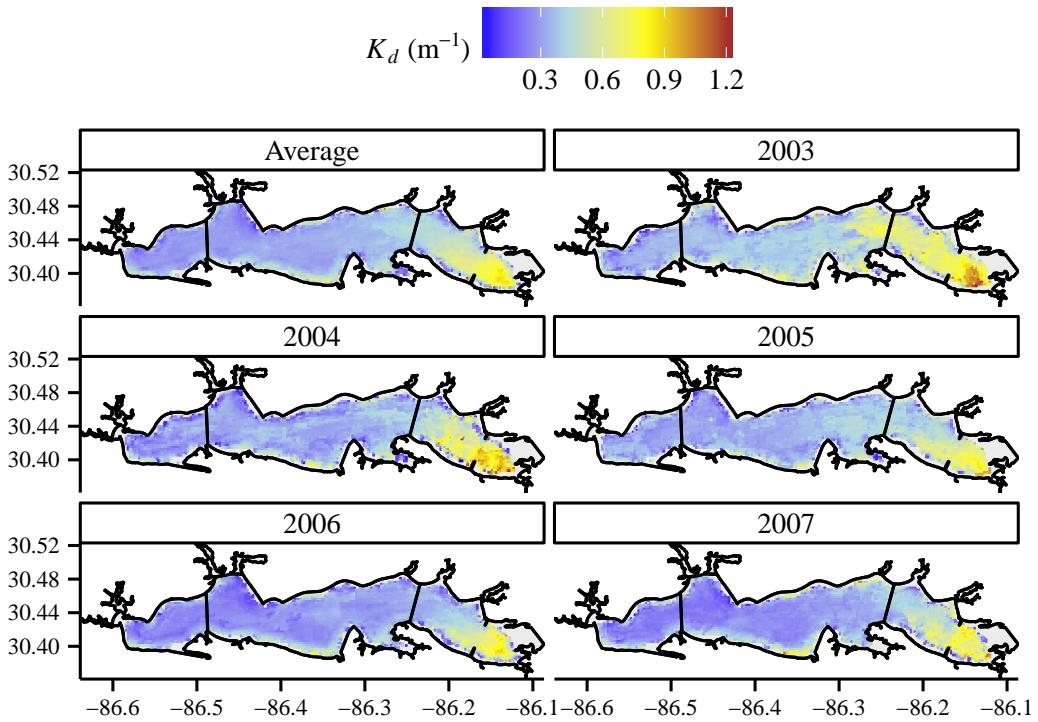


Fig. 5: Satellite estimated light attenuation for Choctawhatchee Bay based on empirical relationships between *in situ* secchi observations and surface reflectance. Each facet is an annual average of light attenuation for available years of satellite data up to the year of seagrass coverage used to estimate depth of colonization. The first facet is an average of all years. See Fig. 7 for segment identification.

{fig:kd_cl}

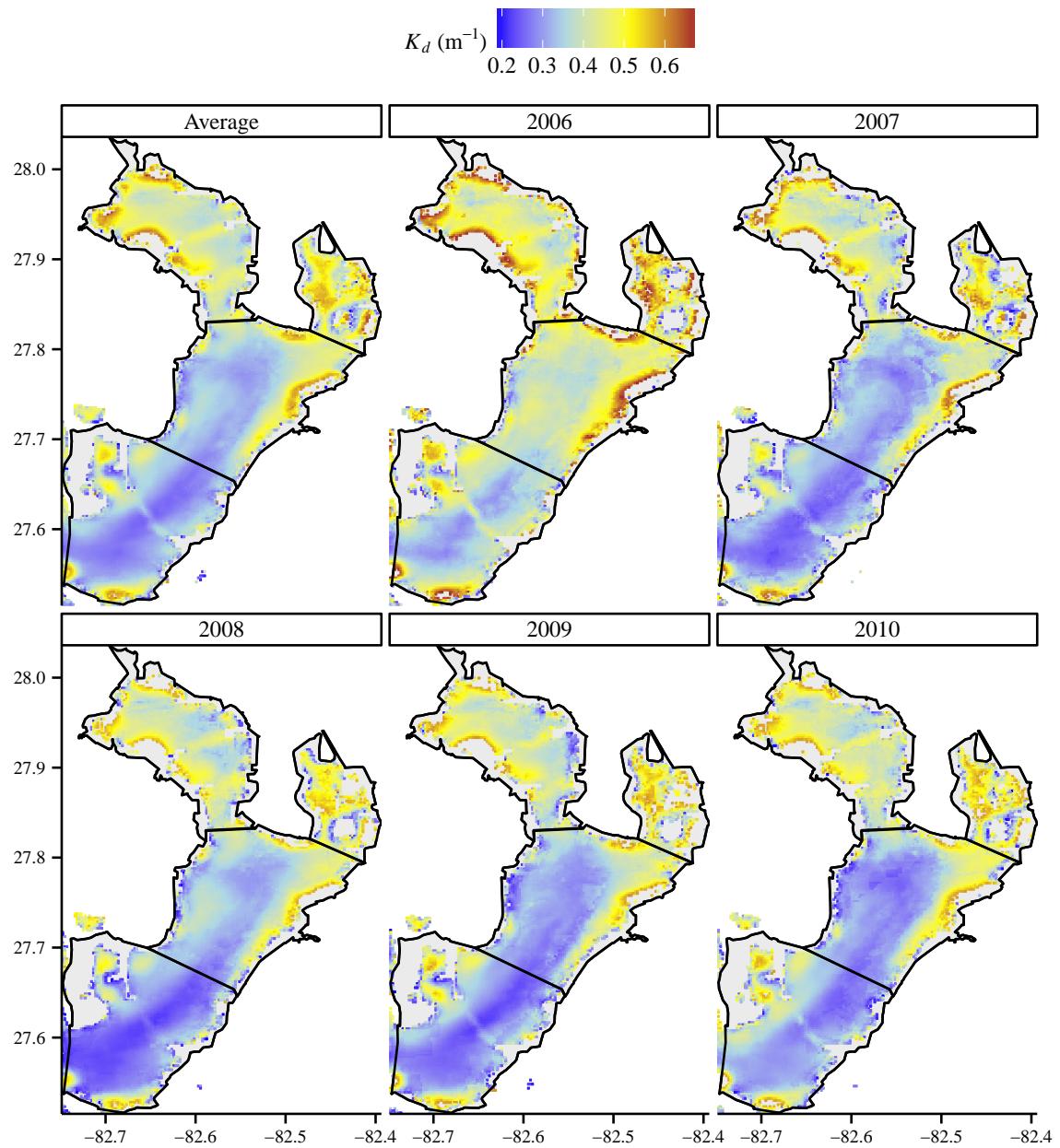


Fig. 6: Satellite estimated water clarity for Tampa Bay based on empirical relationships between *in situ* secchi observations and surface reflectance. Each facet is an annual average of water clarity for available years of satellite data. The first facet is an average of all years. See Fig. 8 for segment identification.

{fig:kd_tk}

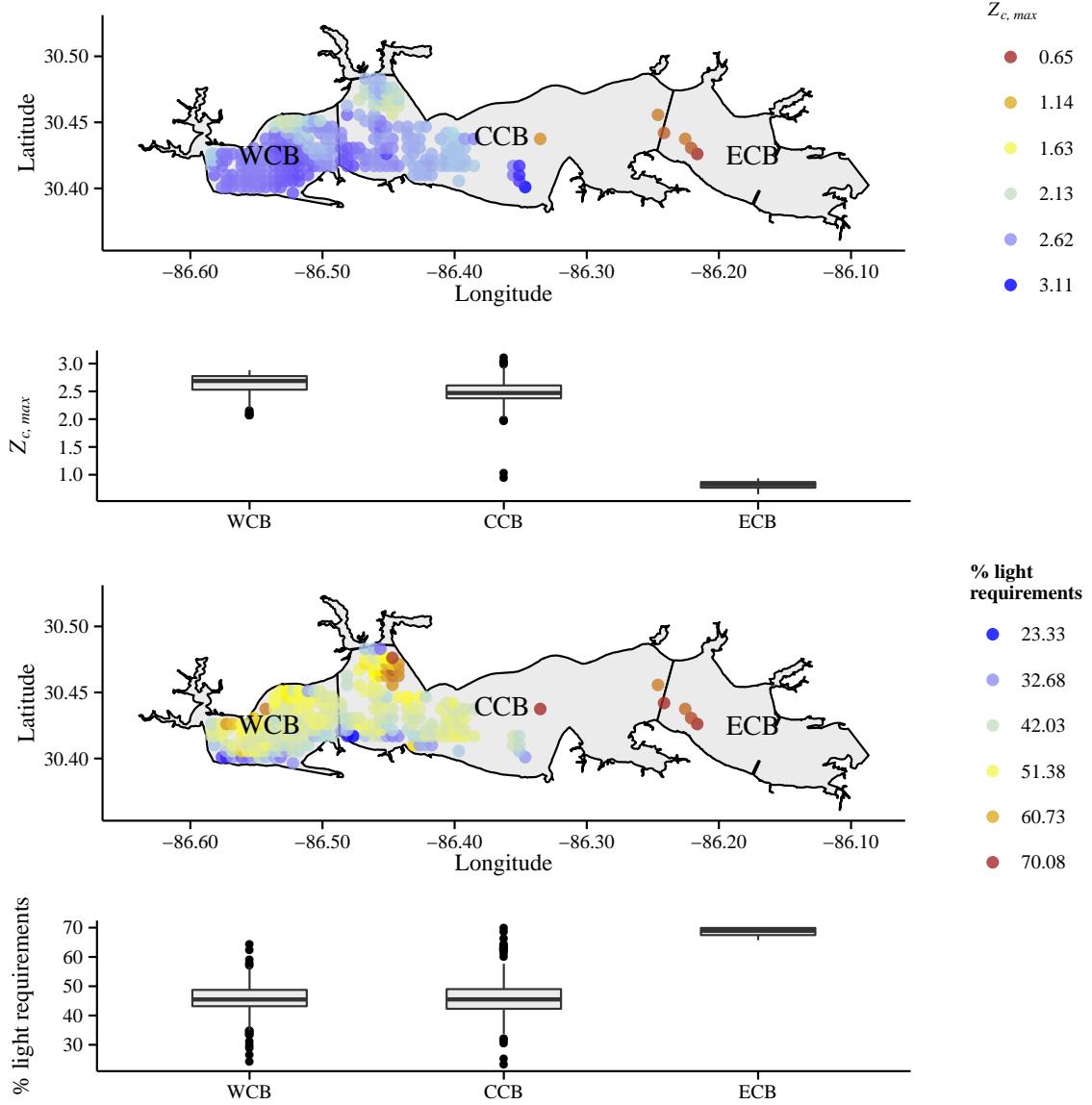


Fig. 7: Estimated maximum depths of seagrass colonization and light requirements for multiple locations in Choctawhatchee Bay, Florida. Locations are those where water clarity estimates were available from satellite observations and seagrass depth of colonization was estimable using a radius of 0.1 decimal degrees. Estimates are also summarized by bay segment as boxplots where the dimensions are the 25th percentile, median, and 75th percentile. Whiskers extend beyond the boxes as 1.5 multiplied by the interquartile range. CCB: Central Choctawhatchee Bay, ECB: East Choctawhatchee Bay, WCB: West Choctawhatchee Bay.

{fig:light}

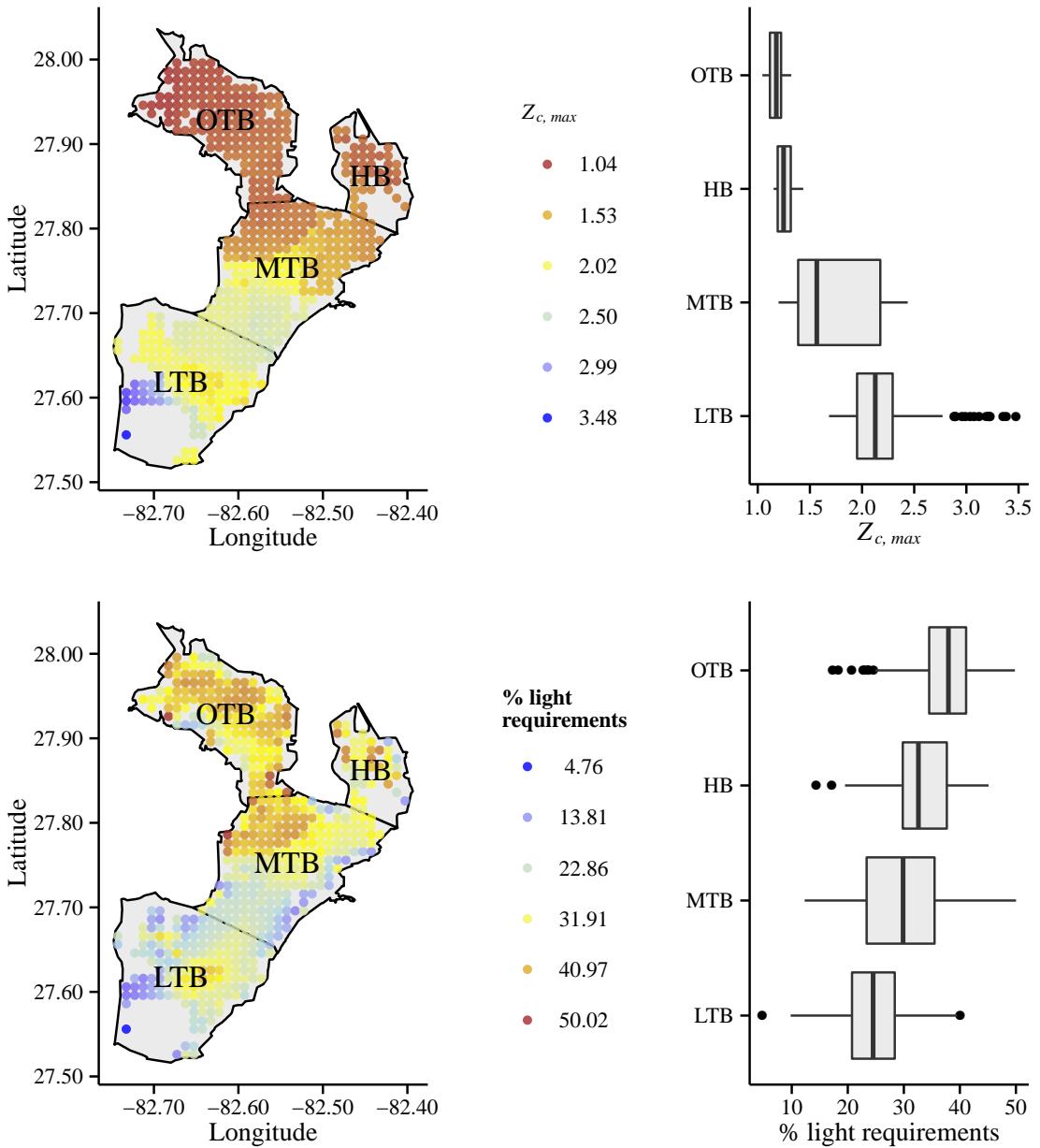


Fig. 8: Estimated maximum depths of seagrass colonization and light requirements for multiple locations in Tampa Bay, Florida. Locations are those where water clarity estimates were available from satellite observations and seagrass depth of colonization was estimable using a radius of 0.1 decimal degrees. Estimates are also summarized by bay segment as boxplots where the dimensions are the 25th percentile, median, and 75th percentile. Whiskers extend beyond the boxes as 1.5 multiplied by the interquartile range. HB: Hillsborough Bay, LTB: Lower Tampa Bay, MTB: Middle Tampa Bay, OTB: Old Tampa Bay.

{fig:light}

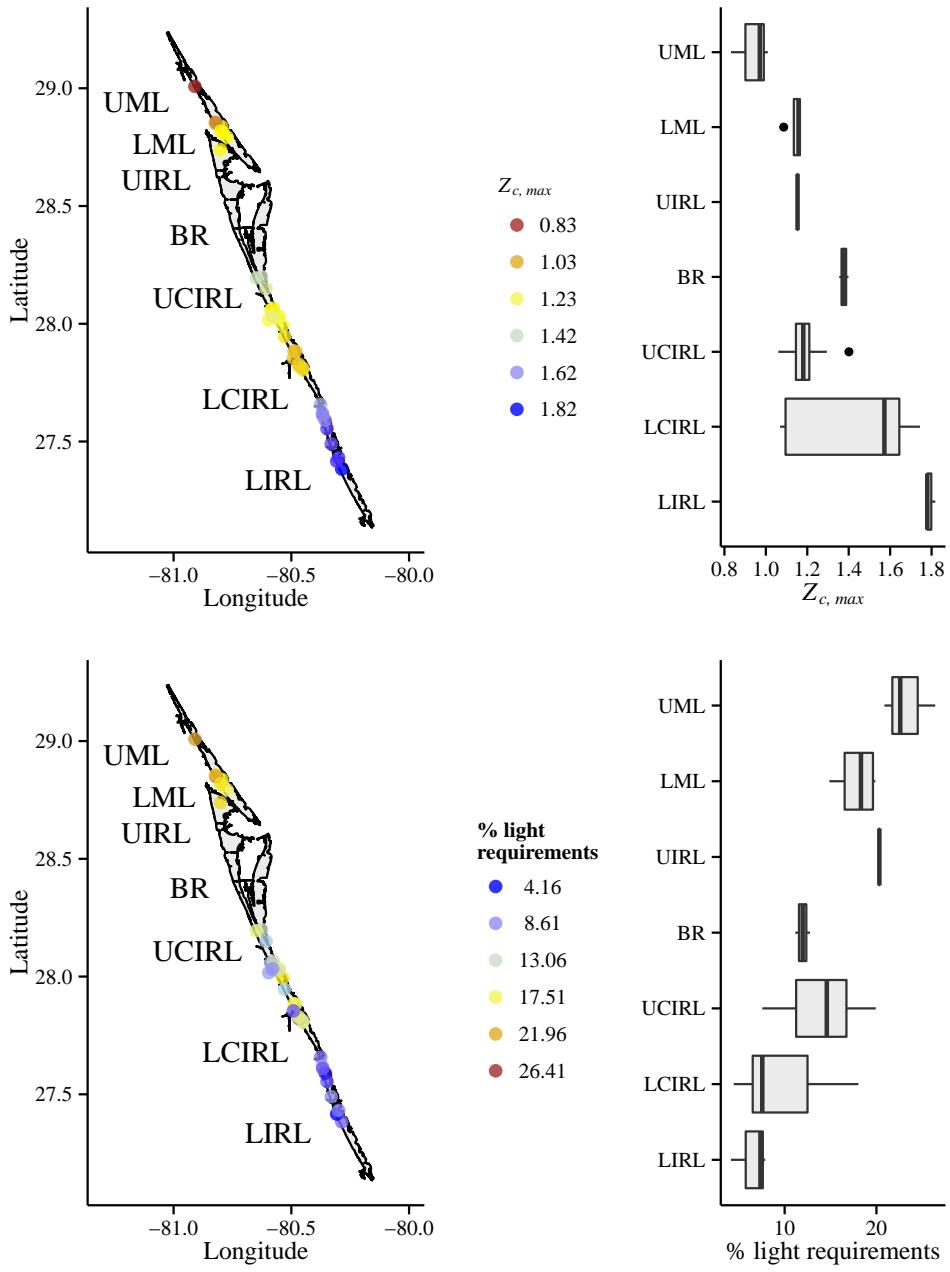


Fig. 9: Estimated maximum depths of seagrass colonization and light requirements for multiple locations in Indian River Lagoon, Florida. Map locations are georeferenced observations of water clarity in the Florida Impaired Waters Rule database, update 40. Estimates are also summarized by bay segment as boxplots as in Fig. 8. Light requirements are based on averaged secchi values within ten years of the seagrass coverage data and estimated maximum depth of colonization using a radius of 0.02 decimal degrees for each secchi location to sample seagrass depth points. BR: Banana R., LCIRL: Lower Central Indian R. Lagoon, LIRL: Lower Indian R. Lagoon, LML: Lower Mosquito Lagoon, LSL: Lower St. Lucie, UCIRL: Upper Central Indian R. Lagoon, UIRL: Upper Indian R. Lagoon, UML: Upper Mosquito Lagoon.

{fig:light}