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Seagrass Community Change at Three High Risk Ports in the Great Barrier Reef World Heritage Area from 2005 to 2014

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Abstract:

Seagrass meadows are extremely valuable and dynamic ecosystems currently facing pressure from anthropogenic disturbances. Seagrass ecosystems are declining globally because of direct and indirect threats that shift environmental conditions controlling seagrass distribution. Seagrass species responses to disturbances vary based on a number of factors including life history strategy. The goal of this study was to map and analyze patterns of dominant seagrass species change at Cairns and Gladstone from 2005-2014 and Townsville from 2007-2014. This compilation data set was symbolized according to the life history strategy of the species. The major disturbances during this time period were physical damage from cyclones and the associated above average rainfall and river flow, which caused large losses in the percent of sites surveyed where seagrass was found. These disturbances shaped the successional patterns observed. Most of the enduring meadows in these ports were composed of the colonizing/opportunistic species, *Z. capricorni and H. uninervis*. Opportunistic and persistent species, such as *C. serrulata* and *T. hemprichii*, were both seen less frequently in these ports. The colonizing species, *H. ovalis and H. decipiens*, were frequently observed colonizing spaced cleared during disturbance. The patterns of succession observed around Cairns, Townsville, and Gladstone support the life-history classifications of the species.

Key Words: GIS, life history strategy, succession, disturbance, and spatial dynamics

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1. Acknowledgements:

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2. List of Figures:

3. Acronyms:

CSIRO……………..…… Commonwealth Science and Industrial Research Organization
ESRI………………… Environmental Systems Research Institute
GIS…………………..Geographic Information Systems
GBRMPA……………Great Barrier Reef Marine Park Authority
GBRWHA…………...Great Barrier Reef World Heritage area
TropWATER………..Centre for Tropical Water & Aquatic Ecosystem Research

4. Introduction:

Seagrass meadows are an incredibly valuable ecosystem in the Great Barrier Reef World Heritage Area (GBRWHA). Seagrass enhances regional biodiversity, carbon sequestration and export, the cycle of nutrients including mitigation of eutrophication, and functions as a nursery or food source for important fauna (Orth et al., 2006). Dugong and green sea turtles are vulnerable (Marsh and Sobtzick, 2015) and endangered species (Seminoff et al., 2004) respectively that rely on seagrass ecosystems in the GBRWHA (Hughes et al., 2009). These highly productive habitats are ecologically linked with similarly valuable coral reefs and mangroves (Grech et al. 2008). The presence of each of these ecosystems benefits the others; for example, seagrass stabilize sediments that could otherwise bury reefs (Orth et al., 2006) and reefs dissipate incoming waves reducing wave energy and turbidity for sheltered seagrass beds (Lowe et al., 2005).

Seagrass meadows are currently declining around the world (Short et al., 2011). These valuable ecosystems face both direct and indirect anthropogenic threats, including eutrophication, sediment run-off, dredging, climate change, aquaculture, and some fishing practices (Orth et al., 2006). These valuable ecosystems are especially susceptible to reduced water clarity because of their high light requirements (Denison et al., 1993). Within the GBRWHA, Grech et al. (2011) used expert knowledge to weight the importance of threats to seagrass meadows. From most to least important, the threats were agricultural run-off, urban/industrial run-off, urban/port infrastructure development, dredging, shipping accidents, trawling, recreational boat damage, commercial boat damage, and netting (Grech et al., 2011). Understanding the relative importance of various threats is important for more efficiently directing management actions.

Seagrass meadows have been described as “coastal canaries” because of their dynamic responses to this diverse slew of influences and disturbances (Orth et al., 2006). The importance of disturbance in shaping community dynamics was identified long ago (Clements, 1916). Disturbance regimes can be either acute (short term with high intensity) or chronic (long term with low intensity) (Tewfik et al., 2007). Coles et al. (2007) presented a similar classification system, but added long-term acute impacts as opposed to just short-term acute impacts. Examples of short-term acute disturbances relevant to the GBRWHA are tropical cyclones and oil spills (Coles et al., 2007). Some long-term acute disturbances are routine dredging operations and frequent boat damage on small scales (Coles et al., 2007). Chronic effects are slow climate change and shifts in outputs of nutrients or herbicides from river systems (Coles et al., 2007). Chronic effects are generally associated with large-scale effects on the dominant composition of meadows, and acute effects may clear space for secondary succession (Tewfik et al., 2007).

The “coastal canary” status implies the state of seagrass meadows is often reflective of the health of marine systems. Seagrass monitoring programs are therefore important for understanding the overall health of marine systems. However, seagrass species have different strategies for growth, survival, recovery, and colonization (Coles et al., 2007). The variation in strategies utilized may influence individuals and meadows responses to natural and anthropogenic disturbance (Coles et al., 2007). Different species have different tolerance ranges for any number of environmental factors such as light, salinity, turbidity, and nutrient levels (Grice et al., 1996, Björk et al., 1999, Longstaff and Dennison, 1999, and Koch et al., 2007). Threats to seagrass have the potential to influence the distribution of seagrass species and therefore, have complex effects on the seagrass community structure (Tewfik et al., 2007).

Seagrass monitoring and management is challenging because of the dynamic nature of seagrass meadows. Kilminster et al. (2015) synthesized a basic classification system for seagrass species to aid managers making decisions with limited resources. Seagrass meadows are either transitory or enduring with enduring meadows existing for five or more years under standard conditions (Kilminster et al., 2015). Seagrasses were grouped into three overarching life-history strategies: colonizing, opportunistic, and persistent (Figure 1) (Kilminster et al, 2015). Colonizing species were named for their fast recovery in the face of disturbance, but they are physically not the best at withstanding disturbance (Kilminster et al, 2015). Persistent species have the opposite traits with high resistance and slow recovery (Kilminster et al, 2015). Opportunistic species have some of the characteristics of both colonizing and persistent species (Kilminster et al, 2015). Colonizing and opportunistic species can form transient or enduring meadows if the conditions are appropriate, but persistent species will mostly form enduring meadows (Kilminster et al., 2015). These seagrass categories are rooted in the trade-off between colonization and competition abilities central to plant ecology (Tewfik et al., 2007).

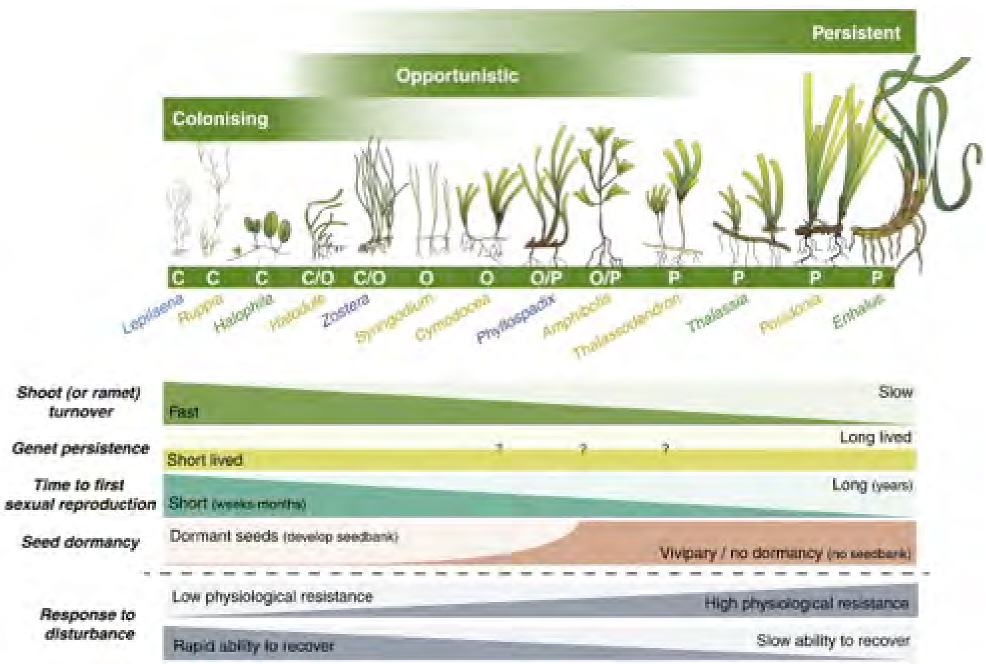

Figure 1: Seagrass genera on the scale from colonizing to persistent and the biological factors that shape the classification. Figure from Kilminster et al. (2015).

This study examines seagrass community change from 2005 to 2014 in Cairns, Townsville, and Gladstone with the goal of analyzing these changes in relation to the seagrasses' life history strategies. These three ports are high-risk zones for seagrass, so disturbances have the potential to occur more frequently there. Each region experienced a period of major seagrass loss within the range of 2009 to 2012. These losses were all connected to a chronic disturbance of above average rainfall and increased river flow (Chartrand et al., 2011, Rasheed et al., 2013, McKenna and Rasheed, 2012). This above average rainfall was associated with a series of tropical cyclones that occurred during this time period: Hamish in 2009, Ului in 2010, Anthony in 2011, and Yasi in 2011 (Carter et al., 2015). These conditions can physically harm seagrasses and their seed banks (Preen et al., 1995). The increased river flow can also influence some of the environmental conditions that affect the ability of seagrass to grow including light, nutrient, and salinity levels (Denison et al., 1993, Orth et al., 2006, and Ralph et al., 2007). Major seagrass losses leave substrate available for secondary succession.

These cyclone and flood related disturbances allow for the examination of succession in seagrass communities following disturbance. Understanding the timescale and patterns of seagrass recovery allows for a better management planning following a disturbance event. The trends in seagrass community change around Cairns, Townsville, and Gladstone should reflect the seagrasses' life history strategies and be shaped by the level and types of disturbance that actually occurred from 2005-2014.

5. Methods:

5.1 Study Site

The GBRWHA spans 348,000 square kilometers of coastal waters from Cape York to just north of Bundaberg (GBRMPA, "About the Reef"). This region was listed as a World Heritage Area, because it is a major biodiversity hotspot. Fifteen seagrass species are found in this region due to the complexity of its coastal habitats (Grech et al., 2008).

5.2 Data Collection Methods

This study uses two GIS data layers on the scale of the GBRWHA. This spatial data is currently not publicly available and permissions to use these layers were granted by the authors A. Carter and A. Grech (Carter et al., 2016, and Grech et al., 2011).

The Carter et al. (2016) layer is a compilation of the TropWATER Seagrass Ecology Group's surveys from 1984 to 2014 and a CSIRO team's similar surveys from 2003 to 2005 (Pitcher et al., 2007, Carter et al., 2016). This ~66,000-point data set includes information such as location, month and year, depth, whether it was growing or senescent season, survey method, Natural Resource Management region, seagrass presence or absence, dominant seagrass species, and seagrass species present from twelve species across three families (Carter et al., 2016). The thirteen species surveyed were *Cymodocea rotundata*, *Cymodocea serrulata*, *Enhalus acoroides, Halophila capricorni*, *Halophila decipiens*, *Halophila ovalis*, *Halophila spinulosa*, *Halophila tricostata*, *Halodule uninervis, Syringodium isoetifolium*, *Thalassodendron ciliatum*, *Thalassia hemprichii*, and *Zostera muelleri* subspecies *capricorni* (abbreviated in this report to *Z. capricorni*). Seagrass assessments were made by boat (video transects, camera drops, free divers, SCUBA divers and van Veen grabs), and from randomly placed quadrats during surveys conducted on foot or from hovering helicopters (Carter et al., 2016). Pitcher et al. (2007) collected seagrass data for a larger seabed biodiversity survey across the GBRWHA. Pitcher et al. (2007) used epibenthic sleds to collect samples and towed video cameras. Carter et al.'s (2016) GIS layer was mapped with symbology showing seagrass presence and absence with color graded to show the age of the data (Figure 2).

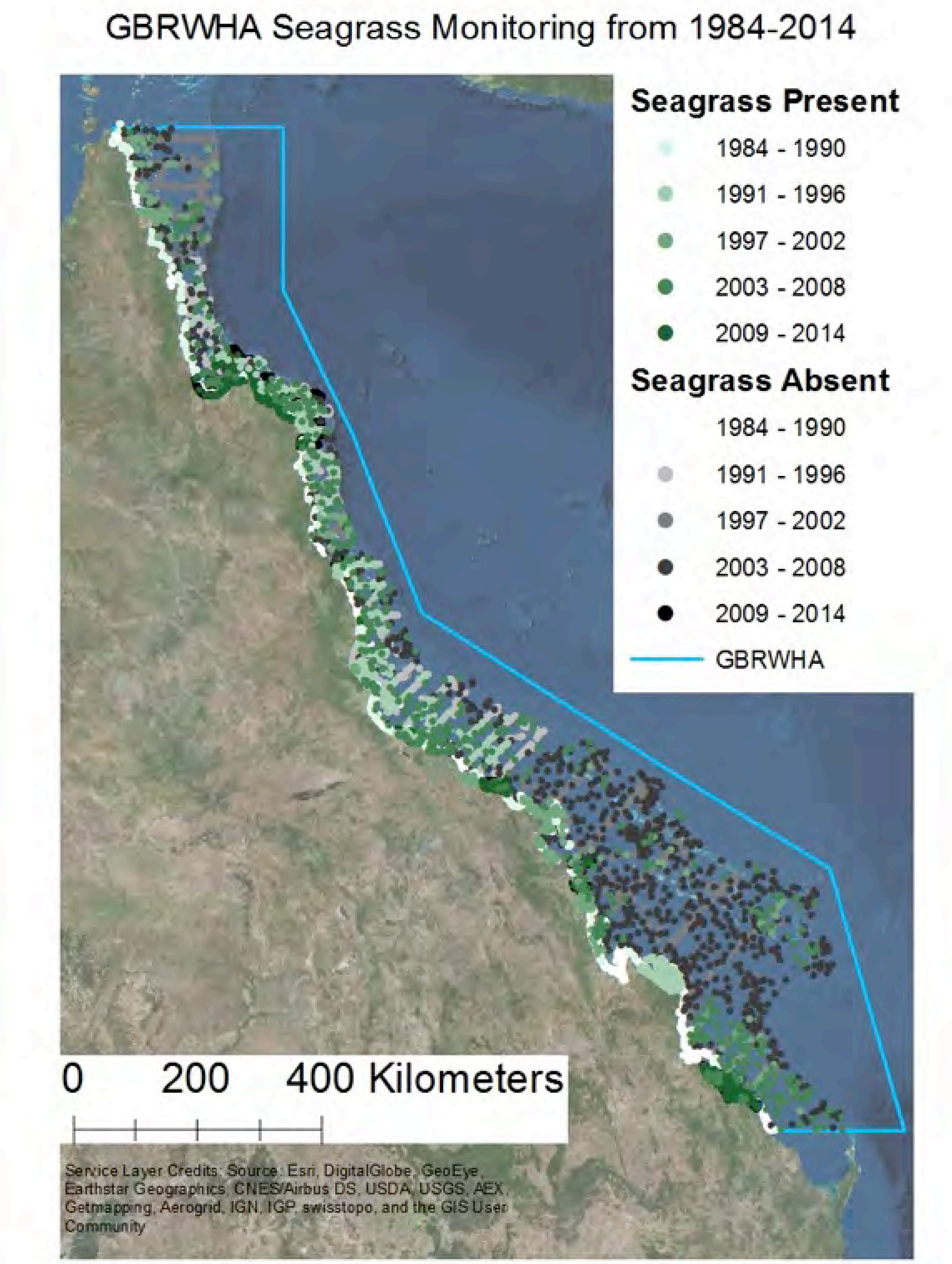

Figure 2: Seagrass presence and absence at survey sites from 1984 to 2014 within the GBRWHA. Seagrass layer courtesy of Carter et al. (2016).

Grech et al. (2011) developed the risk layer used in this study using information gathered in an online survey of seagrass experts. Halpern et al. (2007) designed the survey approach used to determine expert consensus on the comparative impacts of threats to marine systems. The method uses a standardized ranking system for five vulnerability factors including frequency, functional impact, resistance, recovery time, and certainty (Halpern et al., 2007). Grech et al. (2011) gave the vulnerability factors equal weight and combined them into an average vulnerability score. Grech et al. (2011) multiplied these vulnerability scores by zero, a half, or one at specific regions to reflect the amount of the threat occurring in each region. Multiplying by zero means that the threat does not occur there, and multiplying by one means that the threat occurs there at a high level. The locations assigned half of the initial vulnerability score had a moderate level of the threat occurring. The risk layer was created by combining the cumulative threat map with a map predicting where seagrass will occur in the same region (Grech and Coles, 2010). Approximately 10 percent of this coastal region was designated high risk (Grech et al, 2011) (Figure 3).

Figure 3: Predicted seagrass risk levels for coastal waters (≤15m) within the GBRWHA. Risk layer courtesy of Grech et al. (2011).

5.3 Spatial Analysis Methods

Environmental Systems Research Institute's **(**ESRI) ArcMap 10.1 was used for all analysis and map creation in this report. The base map used is courtesy of ESRI.

The seagrass layer was clipped to the risk layer's extent and converted from a point feature to a raster feature. This raster was created using the seagrass presence/absence variable converted to a binary numerical form. The values of the seagrass raster layer were reclassified for the purposes of adding it to the risk layer with the raster calculator tool. The resulting compilation raster had one value for high-risk regions where no seagrass data was collected (Figure 4). This compilation raster was mapped with symbology to reflect the extent of seagrass data collection relative to the location of high-risk areas.

Cairns, Townsville, and Gladstone were identified as high-risk ports with high densities of seagrass data collection. Polygons were created from the high-risk regions around these ports, and these polygons were used to clip the seagrass data. Data from the senescent season of seagrass was removed, so that any seasonal change would not be misinterpreted as change over years. Dominant seagrass species at survey sites were mapped annually from 2005 to 2014 for Cairns and Gladstone and from 2007 to 2014 for Townsville (Figures 5-10). Dominant seagrass species were mapped according to the Kilminster et al. 2015 classification system. This system ranks seagrass species from the long-lived persistent species (lowest level of concern) to colonizing species (highest level of concern). Colonizing species can be of concern because of their role as indicators of disturbance when species composition switches from persistent to colonizing (Kilminster et al., 2015). Trends in community composition change over time were mapped using the time slider and analyzed visually for all three of the ports.

5.4 Limitation of the Data

The seagrass data only included information on presence and absence of different species. This lack of relative biomass data prevented species diversity calculations or any other population comparison metric.

6. Results:

Spatially comparing the extents of the seagrass surveys with the extents of the high-risk regions reveals that from 2011-2014 data has been collected across about thirty percent of the high-risk regions identified in the Grech et al. (2011) layer (Figure 3). Figure 3 was used in the process of determining what locations to focus the analysis on.

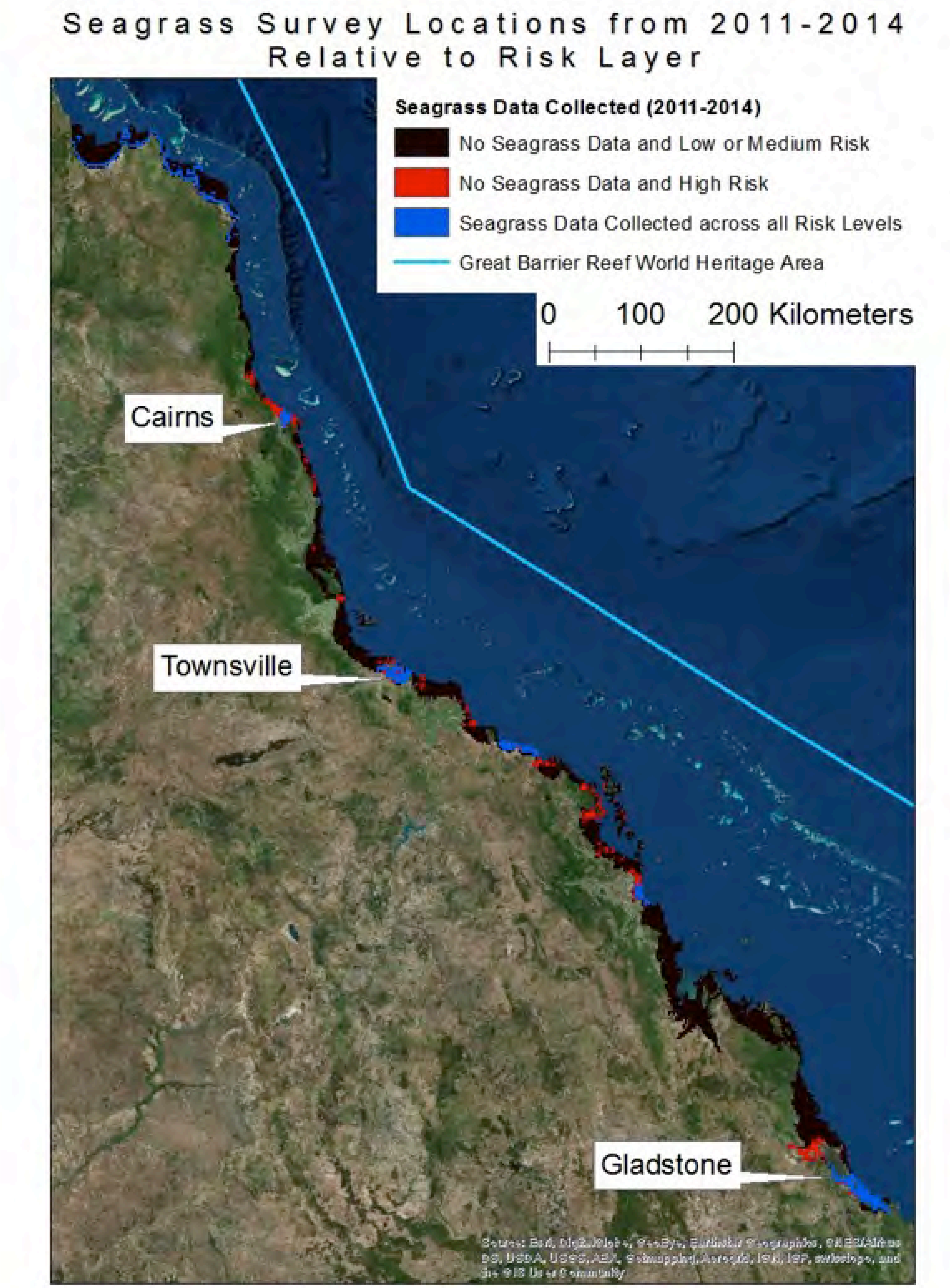

Figure 4: Regions of recent seagrass data collection (2011-2014) relative to low, medium and high risk regions in the GBRWHA.

6.1 Seagrass Community Change around Cairns

In Cairns from 2005-2009, meadow “A” was primarily dominated by *Z. capricorni* and meadow “B” was primarily dominated by *H. uninervis*; these species are both colonizing/opportunistic (Figure 5). *H. ovalis* (colonizing) dominated some of the survey sites along the meadow edges (Figure 5). From 2005 to 2006, *Z. capricorni* replaced *H. ovalis* at the southern end of meadow “B.”

In 2009, 67% of sites surveyed had seagrass. The number of sites with seagrass declined to 2012 when the most dramatic absence of seagrass occurred (Figure 5 and 6). In 2010, 24% of sites surveyed had seagrass. In 2011, only 14% of sites surveyed had seagrass. By 2012, 2% of sites surveyed had seagrass. In 2012, *Z. capricorni* and *H. uninervis* were the only two species dominating in limited regions (Figure 6).

Seagrass recovery and reestablishment occurred in 2013 and 2014 at some regions that were bare in 2012 (Figure 6). The percent of sites with seagrass increased to 9% in 2013, and to 13% of the sites in 2014. *H. uninervis* and *H. ovalis* were the primary species dominating survey sites during this time of recovery (Figure 6). *H. decipiens* (colonizing) was dominant a few times throughout this period of recovery (Figure 6). *T. hemprichii* (persistent) was the dominant species for the only time within these surveys in 2013 (Figure 6).

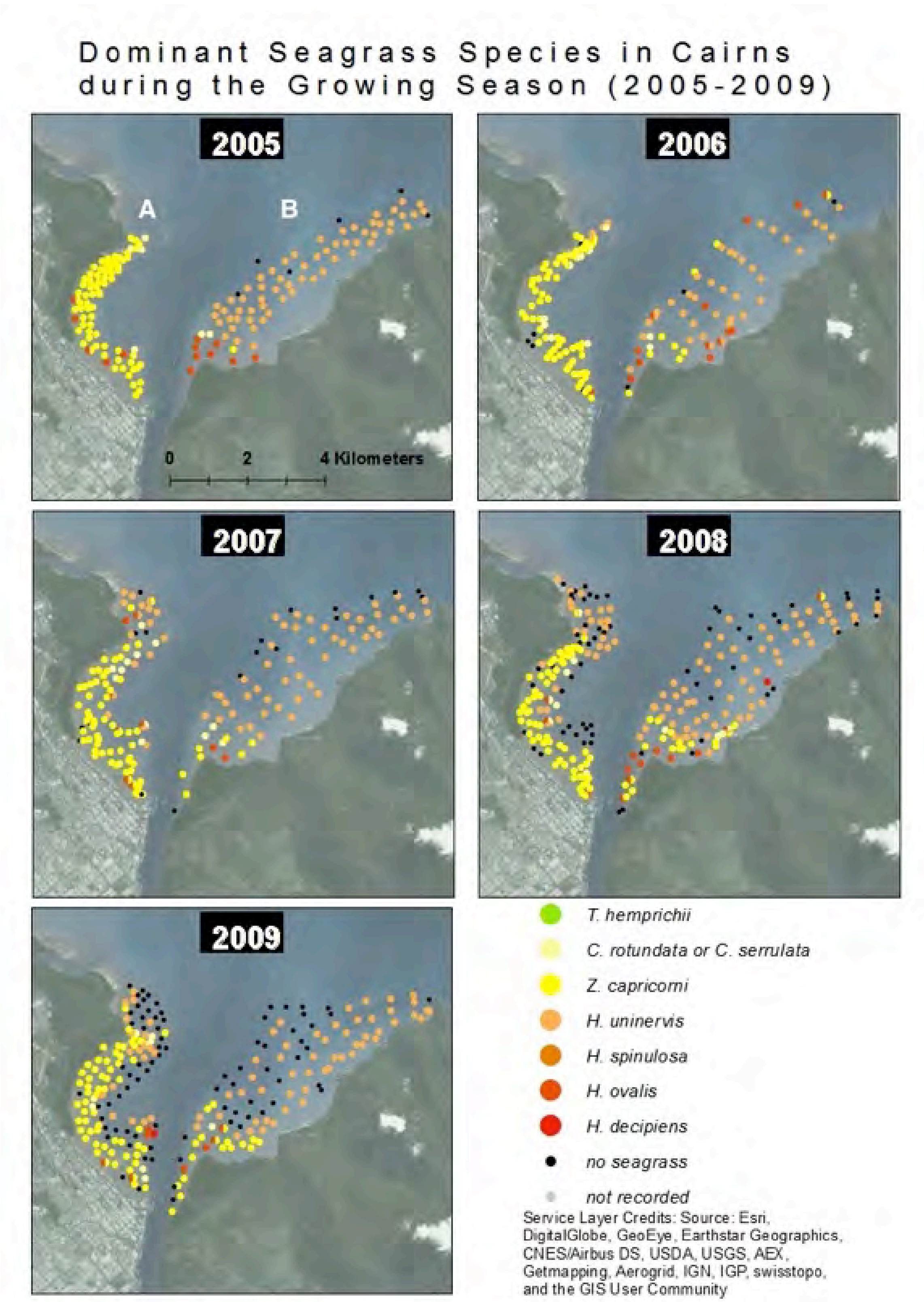

Figure 5: Dominant seagrass species composition in the Cairns region, 2005-2009. A) Western meadow B) Eastern meadow.

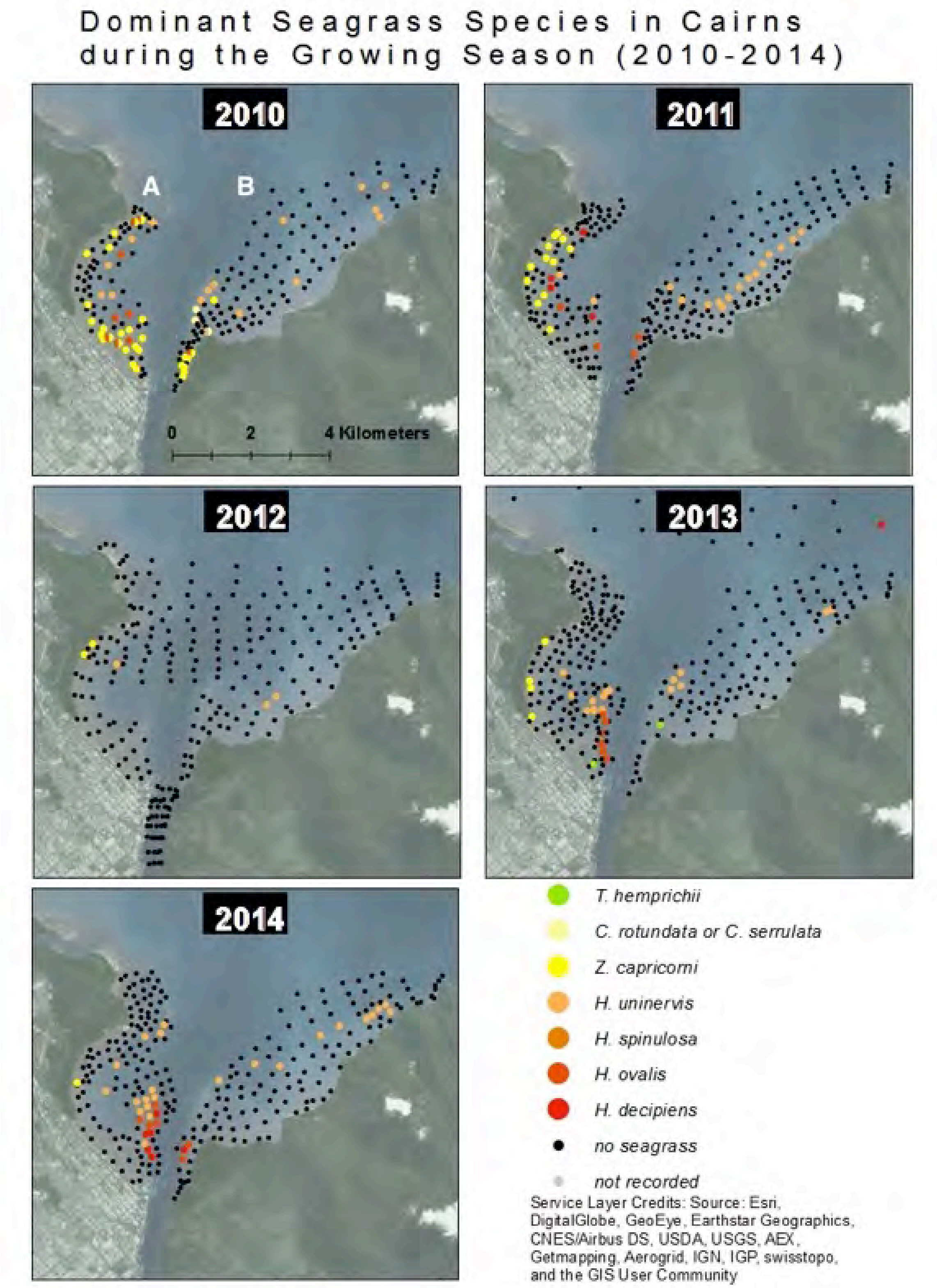

Figure 6: Dominant seagrass species composition in the Cairns region, 2010-2014. A) Western meadow B) Eastern meadow.

6.2 Seagrass Community Change around Townsville

Seagrass communities on the mainland coast near Townsville in meadow “B” from 2007 to 2009 consisted of slightly shifting compositions of *C. serrulata* (opportunistic), *Z. capricorni* (colonizing/opportunistic), *H. uninervis* (colonizing/opportunistic), and *H. spinulosa* (colonizing) (Figure 7). From 2007 to 2009 in meadow “B,” *C. serrulata* was largely replaced by *H. uninervis* and *H. ovalis* (colonizing). Meadow “A” was primarily *H. uninervis* and *H. spinulosa* (Figure 7). From 2007 to 2008, *H. spinulosa* was lost in the region of meadow “A” farther from the coast. In 2009, this region was colonized by *H. uninervis* and *H. ovalis*.

From 2009 to 2010, fewer sites had seagrass, and *H. decipiens* (colonizing) dominant at more sites at meadows “A” and “B” (Figure 7 and 8). In 2009, 59% of sites had seagrass, decreasing to 38% of sites in 2010 and 28% in 2011(Figure 8). In 2011, *H. uninervis* dominated regions of both meadow “A” and “B,” and *Z. capricorni* dominated some regions of meadow “B.”

From 2012 to 2014, the percentage of sites with seagrass increased from 46% to 58%. In 2012, *H. decipiens* colonized some of the regions that were bare on both meadows “A” and “B” during 2011 (Figure 8). In 2013, *H. spinulosa* replaced *H. decipiens* in most of meadow “A.” By 2014, *Z. capricorni*, *H. uninervis*, and *H. spinulosa* recovered in the region (Figure 8).

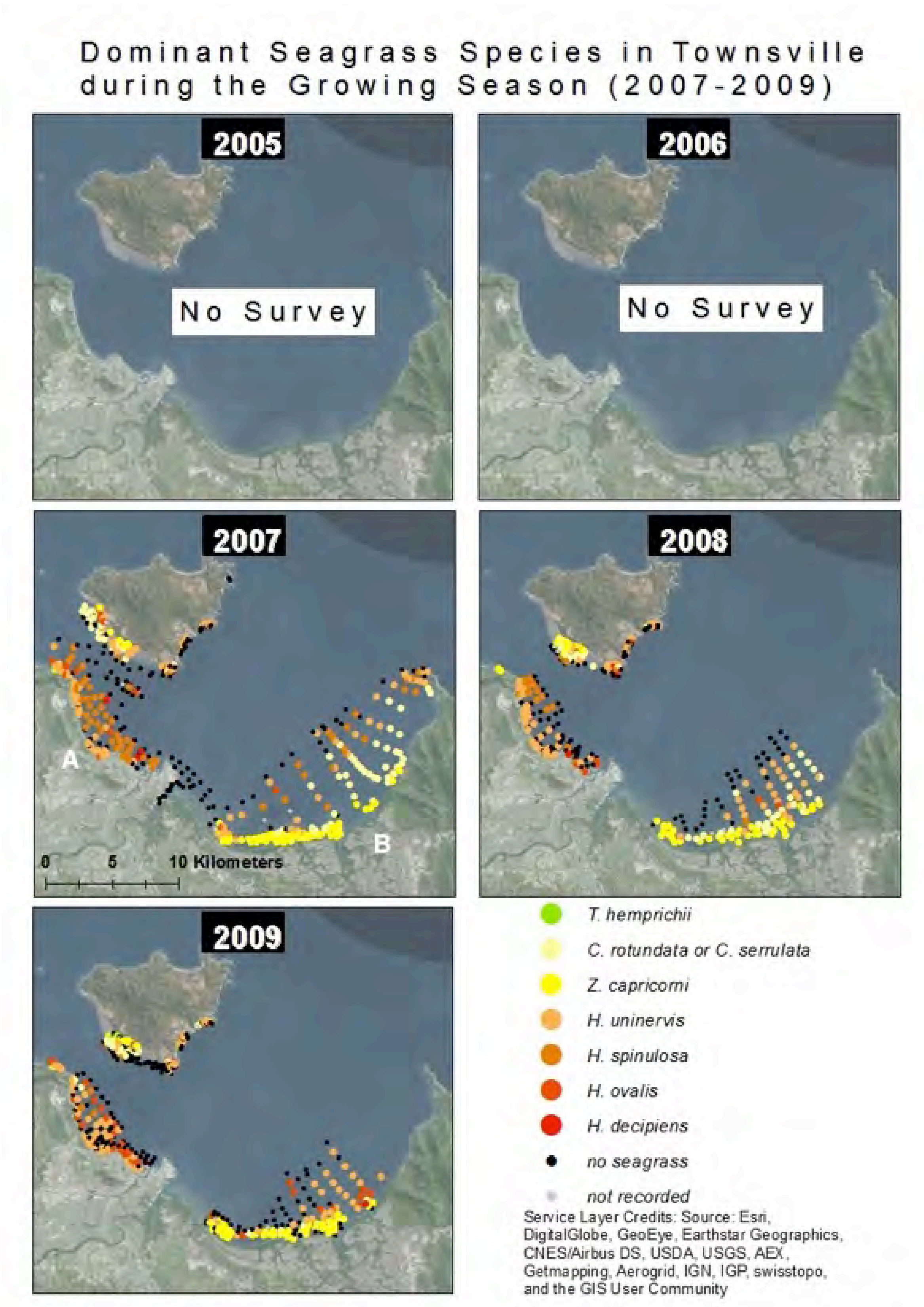

Figure 7: Dominant seagrass species composition in the Townsville region, 2007-2009. A) Western meadow B) Eastern meadow.

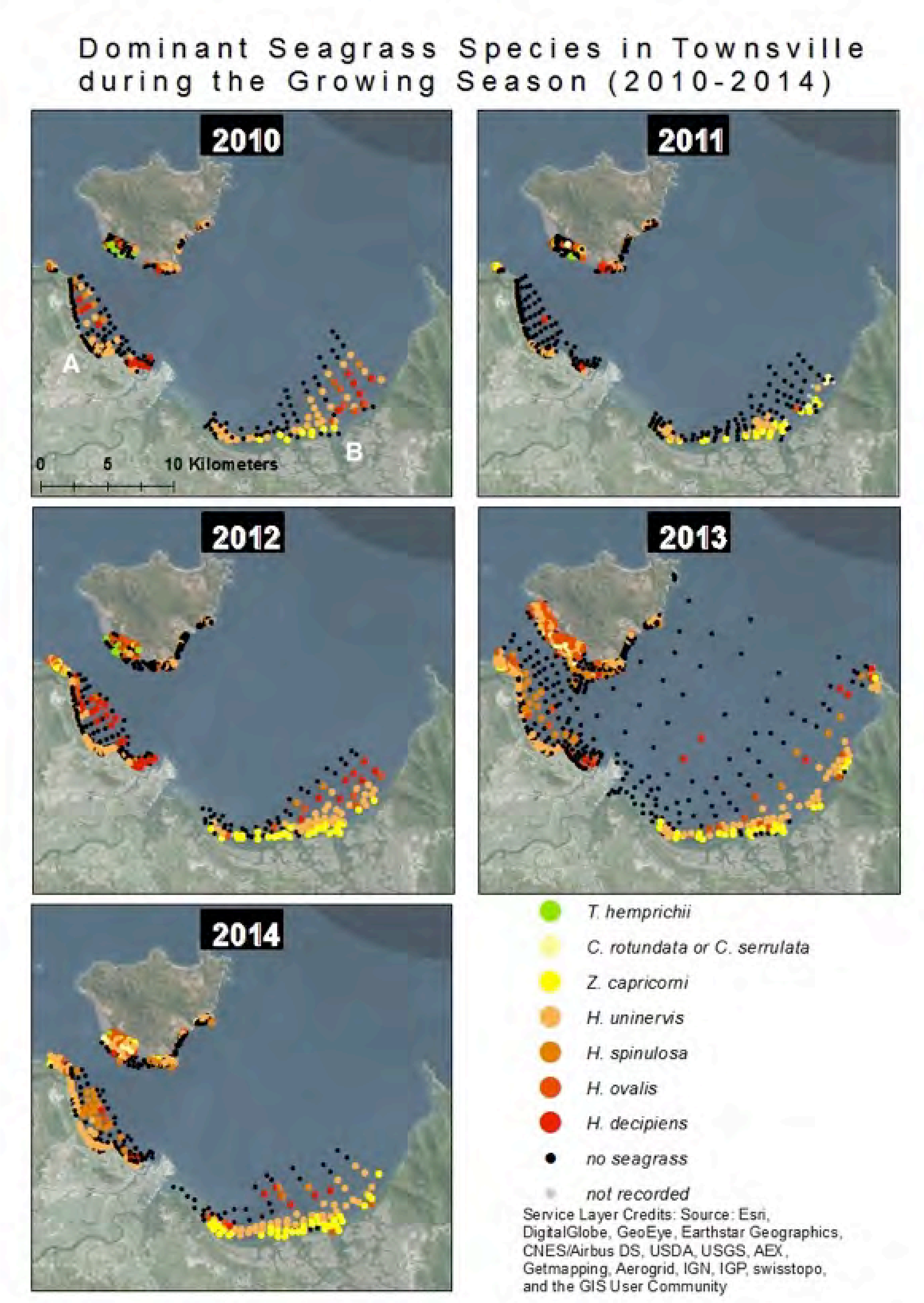

Figure 8: Dominant seagrass species composition in the Townsville region, 2010-2014. A) Western meadow B) Eastern meadow

6.3 Seagrass Community Change around Gladstone

From 2005 to 2009 four main meadows around Gladstone maintained similar dominant species compositions (Figure 9). Meadows “A” and “B” were dominated by a shifting mix of *Z. capricorni* (colonizing/opportunistic), *H. ovalis* (colonizing), and *H. decipiens* (colonizing) (Figure 9). In 2005, there was a section of meadow “A” without seagrass, which was colonized by *H. decipiens* in 2006. Meadows “C” and “D” were dominated by *Z. capricorni* and *H. uninervis* (colonizing/opportunistic) respectively (Figure 9). In 2009, *H. uninervis* was largely lost in meadow “D,” but this year more sampling sites were added and *H. uninervis* was found slightly north of its original distribution. It is unclear whether a distribution shift occurred or whether this new section is a remnant of the initial larger meadow. In 2010, *H. uninervis* recolonized some of meadow “D” from this northern section.

A general loss of seagrass occurred around 2010 in these meadows (Figure 10). In 2009, 40% of sites surveyed had seagrass. This decreased to 25% in 2010 and 2011. Sites with seagrass improved in 2012 (35%), declined again in 2013 to 26%, before increasing in 2014 to 40%.

From 2010 to 2014, *H. ovalis* dominance spread across meadows “A” and “B” previously dominated by *Z. capricorni* and *H. decipiens*; however, *Z. capricorni* persisted in small sections (Figure 10). The *Z. capricorni* dominance endured in meadow “C” with slightly reduced area coverage for the entire time frame (Figure 10). In 2013 and 2014, *Z. capricorni* at the southern end of meadow “C” was replaced by *H. ovalis* (Figure 10). In 2013 and 2014, *H. ovalis* and *H. decipiens* colonized some of the space around the persistent section of *H. uninervis* in meadow “D” (Figure 10).

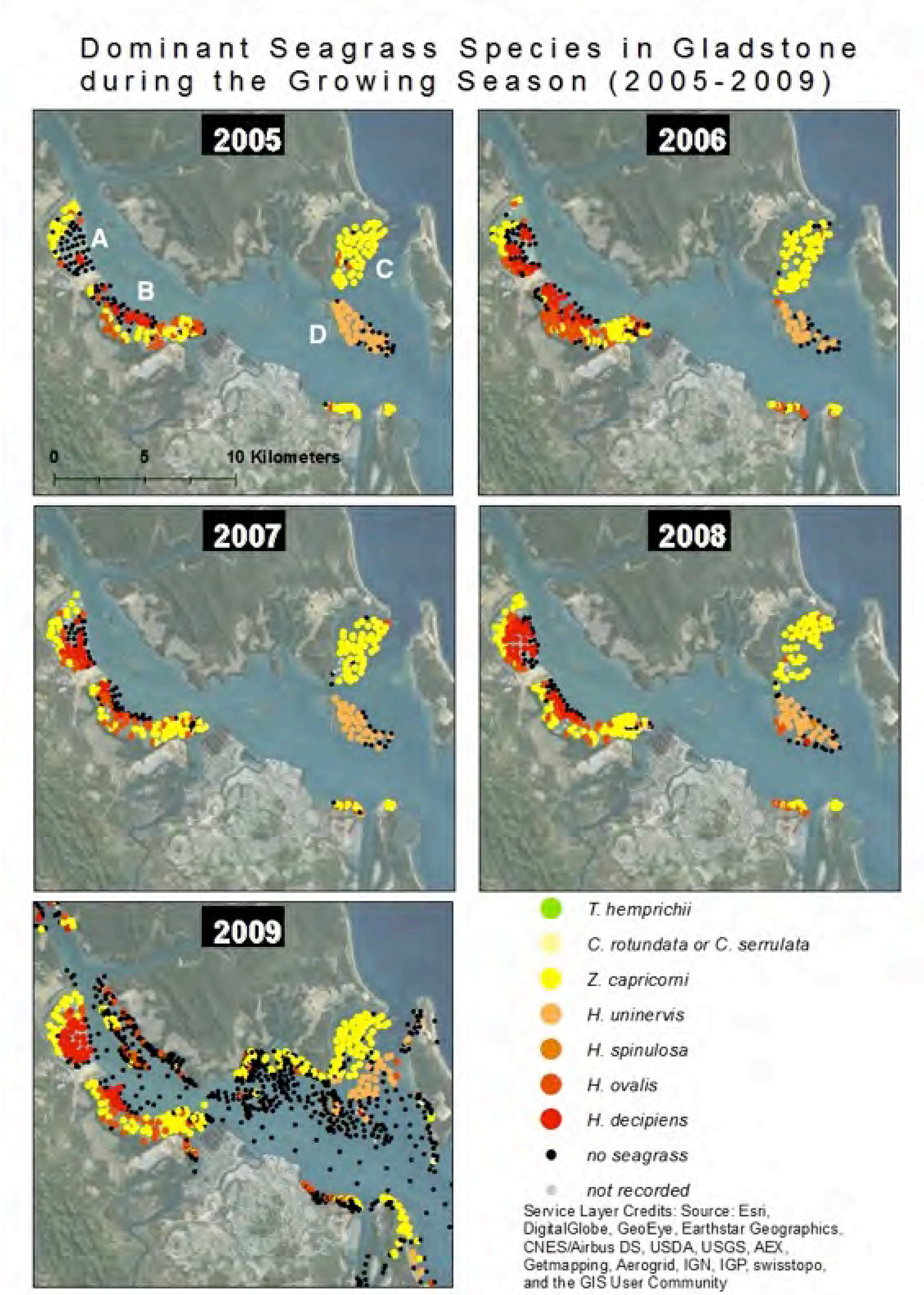

Figure 9: Dominant seagrass species composition in the Gladstone region, 2005-2009. A) Northwestern meadow B) Southwestern meadow C) Northeastern meadow D) Southeastern meadow.

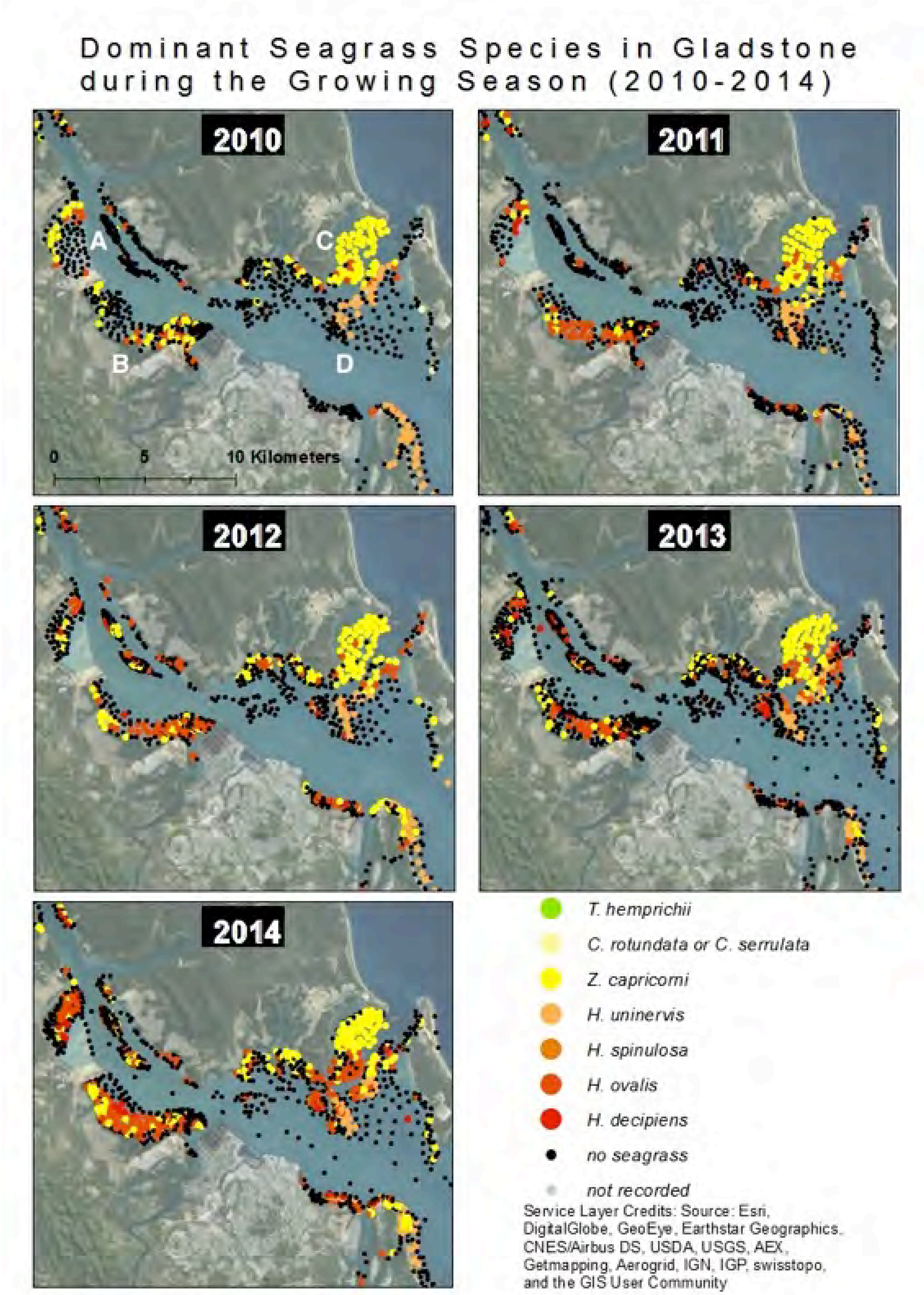

Figure 10: Dominant seagrass species composition in the Gladstone region, 2010-2014. A) Northwestern meadow B) Southwestern meadow C) Northeastern meadow D) Southeastern meadow.

7. Discussion:

The patterns of seagrass community change around Cairns, Townsville, and Gladstone support the life history classifications made by Kilminster et al. (2015) as species closer to the colonizing end of the spectrum dominate following the years of disturbance.

Colonizing and colonizing/opportunistic species were dominant more often than the opportunistic and persistent species in this study, but this may be shaped by factors other than the disturbance level of the ports. Firstly, there are more colonizing species found in tropical North Australia than persistent; there are two persistent, five opportunistic, and eleven colonizing species found in this region (Kilminster et al., 2015). Secondly, all three of these sites are estuaries. Transitory meadows are more likely to occur in highly variable environments like estuaries and persistent species do not often form transitory meadows (Kilminster et al., 2015). Estuaries have this variation because of the variable inflow of freshwater, which shifts the salinity, temperature, and nutrient and sediment levels (Kilminster et al., 2015). It is unclear to what degree long-term anthropogenic disturbance may have had in shifting seagrass community compositions in these estuaries toward the colonizing end of the scale.

Patterns of reestablishment were likely shaped both by life history strategy and the light requirements of the local seagrass species. These two factors are likely linked, because a species would not be an effective colonizing species with exceptionally high light requirements. *Halophila* species are classified as colonizing and require only 10-30% surface light intensity (Freeman et al., 2008). *H. uninervis* is classified as colonizing/opportunistic and also only requires around 10-30% surface light intensity (Freeman et al., 2008). *Z. capricorni* is a colonizing/opportunistic species that is closer to persistent than *H. uninervis*, and *Z. capricorni* requires around 40% surface light intensity (Grice et al., 1996). *Cymodocea* species are opportunistic with a requirement of around 40% surface light intensity (Grice et al., 1996). These trends of life history and light requirement appear to be at least loosely linked, and both drive the successional patterns.

The majority of disturbance that occurred in these ports was a combination of short-term acute with the cyclone damage and chronic with the above average rainfall and increased river flow (Chartrand et al., 2011, Rasheed et al., 2013, McKenna and Rasheed, 2012). The two main disturbance hypotheses for seagrass ecosystems are therefore relevant to the successional patterns that occurred. In the patch dynamic hypothesis, inferior competitors remain in the community through frequent acute disturbance (Paine and Levin, 1981). The inferior competitors are the colonizing species, which are certainly maintained at all three ports. Again, it is unclear how much of this presence of colonizers is associated with the local disturbance regimes and how much is purely a product of the lower numbers of persistent species in tropical North Australia.

In the microhabitat hypothesis, the intensity of chronic disturbance shapes the equilibrium between alternate climax meadow compositions (Tewfik et al., 2007). Discussion of the microhabitat hypothesis involves understanding what the climax communities are in the region studied. In other regions with seagrass like the Caribbean, the climax community is dominated by the most competitively superior species, which is the most persistent species (Tewfik et al., 2007). In these three estuarine ports, the persistent species are rarely the dominant species, so the term climax community may be irrelevant. This hypothesis could be modified to suit these port regions by replacing the term “climax community” with that of just “dominant species.” The chronic disturbance of above average rainfall and increased river flow has a strong shaping force on the successional patterns observed.

The patterns of change around Cairns supports the Kilminster et al. (2015) life-history classification, because the colonizing/opportunistic species do not have high resistance to disturbance and the colonizing species are more quickly able to get reestablished. One specific supporting trend that occurred from 2005 to 2006 was *Z. capricorni* shifting to dominate a region previously dominated by *H. ovalis*. Colonizing/opportunistic species should be superior competitors to colonizing species (Kilminster et al., 2015), so in a habitat where both species types are present, the colonizing/opportunistic species should be able to take over the dominant position. Another specific trend is that *H. uninervis, H. ovalis* and, *H. decipiens* were the main species colonizing after the major 2012 disturbance. *H. uninervis* more successfully rebounded than *Z. capricorni* and is closer to the colonizing end of the scale than *Z. capricorni*. Another potential reason is that *H. uninervis* has lower surface light requirements than *Z. capricorni* (Grice et al., 1996, Freeman et al., 2008).

One trend at Cairns seems contradictory to the life history categories. The dominance of *T. hemprichii* at two locations in 2013 where it was absent from 2005-2012 and later in 2014 is puzzling, as *T. hemprichii* is a persistent species. There are a few possible explanations for this presence. *T. hemprichii* could have been missed in previous surveys and survived through the cyclone damage in 2012, but it seems unlikely that it would have been missed for seven years in a location that was continually sampled. *T. hemprichii* is known for having seeds with little to no dormancy capability (Inglis, 2000), so it seems unlikely that this species was able to reestablish from any sort of seed bank. It is possible that these meadows were able to reestablish from dispersion of propagules, fruits, or fragments from nearby meadows that were able to withstand the disturbance in 2012.

Successional change in seagrass meadows around Townville from 2007-2014 mostly functioned under the expected patterns. During periods of low seagrass presence at sites before and after 2011 the colonizing species, *H. ovalis* and *H. decipiens*, became more common. Townsville experienced faster meadow recovery in 2012 than any other port monitored during the loss period from 2009-2012 (Davies et al., 2013). The process that happened could be described as a quick successional loop, because the space without seagrass in 2011 was largely filled by colonizing species in 2012; by 2014, the seagrass community around Townsville had returned to communities similar to that from 2007-2009 minus the *C. serrulata*. The colonizing/opportunistic species common to this region and *H. spinulosa* (the most persistent of the colonizing species) were able to ultimately outcompete *H. decipiens*.

One occurrence against trend was that, from 2007 to 2009, the opportunistic species, *C. serrulata,* lost its dominant meadow space to *H. uninervis* and *H. ovalis*. It is unclear whether this is the result of loss due to disturbance or the result of being outcompeted. One might expect that an opportunistic species would be more resilient to disturbance than colonizing/opportunistic species with the same surface light requirement. *C. serrulata* and *Z. capricorni* have roughly the same light requirement (Grice et al., 1996), but *Z. capricorni* was more resilient in the community around Cairns than *C. serrulata*. The reason for this difference in ability to withstand disturbance is unclear, and may involve another physiological feature. This suggests that the ranking from persistent to colonizing should not be interpreted too rigidly, and the division between a colonizing/opportunistic and opportunistic species is not a hard line. Trends of resilience and succession following disturbance can be influenced by mechanisms or habitat characteristics that are not immediately obvious to observers (Tewfik et al., 2007).

The patterns of seagrass change around Gladstone are logical in the context of the life histories of the seagrass species present. *Z. capricorni*, as a colonizing/opportunistic species, was the species closest to the persistent end of the life history scale present around Gladstone. Therefore, *Z. capricorni* would be the species predicted to best withstand disturbance. From 2005-2014, *Z. capricorni* did have the most consistent distribution. *H. uninervis* is also a colonizing/opportunistic species, but it is closer to colonizing than *Z. capricorni*; from 2005-2014, there were more fluctuations in the dominance of *H. uninervis*. When seagrass was lost around Gladstone, the colonizing species, *H. ovalis* and *H. decipiens* were most often the species to utilize the space in the following years.

One of the major limitations of this study is that there are no control sites, so comparisons could not be made about successional patterns in high-risk regions and low-risk regions. Long-term seagrass monitoring has not occurred in a low risk estuarine region. This could help determine if direct anthropogenic effects influenced any of the patterns, as a majority of the large disturbances were associated with cyclones. Since the major disturbances were cyclones, the risk level of these regions does not really play an obvious role in the patterns of change as where a cyclone damages most severely is unrelated to the risk level. One potential sources of error is that the annual variation in the survey area may have caused obscured some of the visual patterns and confounded the comparisons of the percent of sites with seagrass for each year.

8. Conclusion:

Seagrass succession around Cairns, Townsville, and Gladstone from 2005-2014 generally followed the expected patterns with knowledge of the life history strategies. The primary implication of this study is that the Kilminster et al. (2015) synthesis of life-history data is well supported and should be applied by the seagrass monitoring and management communities. It will be valuable to continue seagrass monitoring efforts to build an even better idea of how seagrass will respond to disturbance as the intensity and number of tropical storm systems may increase with climate change (Knutson et al., 2010).

One potential direction for further research would be to monitor a region likely to experience disturbance for both seagrass and macroalgae to get a more complete picture of successional patterns. Dense areas of macroalgae function as an alternate climax community in Caribbean macrophyte beds (Tewfik et al., 2007). The data used in this study includes no macroalgae information, so this piece of the puzzle is missing when attempting to understand successional patterns.

As a majority of the damage to seagrass ecosystems that occurred in the examined time period was from tropical cyclones, the most effective way to reduce seagrass loss may be to manage carbon emissions. The direct management actions are still incredibly important if seagrass is going to have a chance at being resilient to anthropogenic climate change. Direct management of seagrass ecosystems involves controlling inputs into watersheds, preventing or minimizing impact from dredging and boats, and restricting development of the ports. Based on the dominant disturbance regimes around these ports from 2005-2014, the mitigation of climate change and laws to protect seagrass directly are both necessary to insure the future of this valuable ecosystem.

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