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The Effects of Sampling Design on Abundance and Distribution of Bottlenose Dolphins in the St. Johns River, Florida

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THE EFFECTS OF SAMPLING DESIGN ON ABUNDANCE AND DISTRIBUTION OF
BOTTLENOSE DOLPHINS IN THE ST. JOHNS RIVER, FLORIDA

by

Samantha Ryanne Nekolny

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in partial fulfillment of the requirements for the degree of

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CERTIFICATE OF APPROVAL

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Abstract

The National Oceanic and Atmospheric Administration (NOAA) is tasked with upholding the regulations prescribed in the Marine Mammal Protection Act of 1972. One of the ways this is accomplished is by producing regular stock assessment reports. For these reports, knowledge about a species abundance and distribution is a vital component, allowing informed management decisions to be made that may potentially reduce the natural and anthropogenic impacts on the organisms. In 2010, an Unusual Mortality Event (UME) occurred in the lower St. Johns River, resulting in an unusually large number of bottlenose dolphin strandings. As the data regarding this population was outdated, the cause of the UME could not be positively determined. Consequently, this lack of information provided the impetus to focus research on the individuals within the lower St. Johns River. Chapter one documents the abundance and emigration rates of the bottlenose dolphin population within the St. Johns River in order to provide NOAA with the necessary information to update the stock assessment report for this population. To designate individuals to various stocks, their distribution must first be known. However, many previous home range studies limited the size of the study area surveyed, which resulted in underestimates of the home range sizes. Therefore, in chapter two I investigated the effects that study area size has on the home range estimates of bottlenose dolphins while demonstrating the usefulness of collaborative science. Together these two chapters report data on the bottlenose dolphin population within the St. Johns River that has not been studied in over 16 years and this research also reveals how study design impacts various analyses and perceived outcomes.

Introduction

The Marine Mammal Protection Act (MMPA) was enacted in 1972 to protect all marine mammals by limiting the “take” of these animals within United States waters. To carry out this protection, the National Oceanic and Atmospheric Administration (NOAA) is tasked with monitoring these species and determining when they fall below the optimum sustainable level (Moore & Merrick 2011). To do this, NOAA assigns individuals to various stocks and prepares stock assessments for all stocks within the United States (Waring et al. 2010). Stocks are defined as groups of a species in the same spatial area that interbreed with one another (U.S. Department of Commerce 1985). Stock assessment reports are updated every three years and every year for stocks classified as “strategic” under the MMPA (Waring et al. 2010). Within these stock assessment reports, abundance estimates for each stock are typically included, with which the potential biological removal (PBR) level can be calculated. The PBR level is the number of individuals that can be taken while still maintaining viable population levels (NOAA Fisheries 2012). Therefore, since abundance estimates are the baseline for protecting these species, care must be taken to ensure that these estimates are accurate. Along with obtaining accurate abundance estimates, the designation of individuals to stocks must also be appropriate. To determine which stocks specific individuals belong to, one must first obtain data regarding the species’ distribution.

On the east coast of the United States, bottlenose dolphins (*Tursiops truncatus*) have been found to range from Long Island, NY to the Florida Keys, FL (Wang et al. 1994). From 1995-2001, NOAA classified all coastal bottlenose dolphins along the western North Atlantic as a single migratory stock (Cupka & Murphy 2005). In 2010, NOAA separated this single stock into five individual coastal stocks (Waring et al. 2010). It was not until 2009 that NOAA began to

recognize Atlantic estuarine stocks (Waring et al. 2009). To date there have been 7 estuarine stocks identified along the east coast of the United States: Northern North Carolina Estuarine System stock, Southern North Carolina Estuarine System stock, Charleston Estuarine System stock, Northern Georgia/Southern South Carolina Estuarine System stock, Southern Georgia Estuarine System stock, Jacksonville Estuarine System stock, and Indian River Lagoon Estuarine System stock (Waring et al. 2009). Genetic analyses are often utilized to determine demographic separations, but in cases where this is not available, data regarding movement and ranging patterns are analyzed. However, the movement and ranging data that is used to designate the stock boundaries is not typically designed for that specific purpose, and therefore the stock boundaries can often times be arbitrary. In many cases, the stock boundaries may not reflect the actual boundaries observed by the animals.

The two recognized estuarine stocks of bottlenose dolphins along the east coast of Florida are the Jacksonville Estuarine System (JES) stock and the Indian River Lagoon Estuarine System (IRLES) stock. The JES stock is designated as the individuals inhabiting the area from the Florida/Georgia border south to Jacksonville Beach, FL (NOAA Fisheries 2009b). To the south is the IRLES stock, which is classified as the individuals ranging from Ponce de Leon Inlet to Jupiter Inlet in south Florida (NOAA Fisheries 2009a). While the IRLES stock has received much attention, the JES stock has received relatively little. Current information on the JES stock is lacking, as the last study on these individuals was conducted from 1994-1997. An Unusual Mortality Event (UME) occurred in the lower St. Johns River (SJR) in 2010 and resulted in a total of 17 dolphins stranding (Jacksonville University & University of North Florida 2011). In response to this UME, research has recently been focused on the SJR portion of the Jacksonville

Estuarine System stock. As information regarding the status of these individuals was outdated, the cause and effects of this UME could not be clearly determined.

Along with evaluating health, natural and anthropogenic impacts from large-scale habitat changes can also be assessed if there is quality baseline data regarding the habitat and its organisms prior to the event. For example, the U.S. Army Corps of Engineers has recently received approval to conduct a harbor expansion project, deepening the main channel of the SJR by over 2 m (Bostick 2014). The purpose of this dredging operation is to increase shipping traffic in an already heavily industrialized area with considerable shipping and boat traffic. Impacts such as this may have a substantial effect on the bottlenose dolphin population within the river as they are thought to have a potentially small population size (NOAA Fisheries 2009b). Therefore, current data on their abundance and distribution is needed. Hence, in order to fill that data gap, the University of North Florida started conducting weekly photo-identification surveys on the dolphins within the SJR in March 2011.

As research regarding the SJR dolphins had not been conducted in over 16 years, these following chapters work to update that information. Chapter one indicates that a large number of dolphins inhabit the SJR and that there is year-round utilization within the SJR, which has a number of management implications. Chapter two indicates that an accurate understanding of dolphin ranging patterns cannot be assessed at a small scale (except possibly for core areas) and illustrates the importance of collaborative science. Together, these two chapters provide essential data that will enable NOAA to update management plans for the JES stock and assist other researchers in planning the sampling designs for their studies of abundance and distribution.

Chapter 1

Mark-recapture abundance estimates of bottlenose dolphins (*Tursiops truncatus*) in the St. Johns River, Florida: a comparison of two sampling designs

Abstract

Marine mammal management decisions are often based on population abundance estimates, yet the St. Johns River (SJR; Jacksonville, FL) bottlenose dolphin community has not been assessed in over 16 years. To address this data gap, vessel based photo-identification surveys were conducted in a 40 linear km study area within the SJR using two study designs. The first approach consisted of a full year of weekly surveys and the second approach utilized less frequent seasonal surveys for three years. Based on the discovery curve for the full year time period, the identification of new individuals appears to have leveled off with a total of 261 individuals identified, not including calves. In contrast, the discovery curve for the three-year time period indicates that the identification of new individuals was steadily increasing with a total of 200 individuals identified, not including calves. Data from both time periods, which were collected and processed using mark-recapture methods, were analyzed in the program MARK to estimate abundance and temporary emigration rates. The robust design with a Markovian temporary emigration model for a semi-closed population was applied. Estimates from the full year approach varied from 160 (95% CI=151-177) in winter to 250 (95% CI=243-263) in summer.

The highest probability of transition to an unobservable state was estimated to occur between summer and autumn, when transients and seasonal residents emigrate out of the area. The abundance estimates from the three-year approach varied from 67 (95% CI=60-87) in winter 2013 to 210 (95% CI=154-316) in summer 2011. The estimates for the first year were the highest compared to the other two years. Emigration was estimated to be greatest between summer and winter across all years. These estimates are the first assessments of bottlenose dolphin abundance within the SJR community alone and provide valuable baseline data for assessing future anthropogenic impacts.

Introduction

Along the east coast of the United States, bottlenose dolphins (*Tursiops truncatus*) are widely distributed from Long Island, New York to the Florida Keys, and inhabit offshore waters, nearshore coastal waters, and estuarine systems (Wang et al. 1994). The National Oceanic and Atmospheric Administration (NOAA) is tasked with managing these populations. Currently, all management decisions are based on the potential biological removal (PBR) level, which is defined as the maximum number of individuals that can be removed while still maintaining viable population levels (NOAA Fisheries 2012). Since accurate abundance estimates are crucial for calculating the PBR level (NOAA Fisheries 2012), abundance and distribution data are vital in making informed decisions regarding the management of bottlenose dolphin populations. Although many of the coastal and estuarine bottlenose dolphin populations inhabiting the eastern United States have been well-studied (Gubbins 2002; Zolman 2002; Torres et al. 2005; Mazzoil et al. 2008; Laska et al. 2011; Toth 2011), current knowledge of population characteristics is lacking for some estuarine areas in Northeast Florida.

The Marine Mammal Protection Act requires that the National Marine Fisheries Service provides stock assessment reports of all marine mammals within the United States every three years and annually for strategic stocks. NOAA currently recognizes two estuarine stocks of bottlenose dolphins along the east coast of Florida: the Jacksonville Estuarine System (JES) stock (NOAA Fisheries 2009b) and the Indian River Lagoon Estuarine System (IRLES) stock (NOAA Fisheries 2009a). In comparison to the IRLES, which has been studied extensively (Mazzoil et al. 2008; Durden et al. 2011), the current stock assessment report for JES is based on data that is 16 years old. Prior information on the Jacksonville Estuarine Stock was limited to a study conducted from 1994-1997, which utilized behavioral and genetic data to conclude that there were three separate bottlenose dolphin communities in Jacksonville: northern, southern, and coastal (Caldwell 2001). Assemblages of individuals that socialize more often with one another than with other individuals were termed communities (Wells 1986). The southern community was defined as the individuals within the St. Johns River (SJR) plus 2 km of the adjacent intracoastal waterway to the south (Caldwell 2001). Currently, bottlenose dolphins within the southern (SJR) community are considered to be part of the JES stock, which spans from the Florida/Georgia border at Cumberland Sound to Jacksonville Beach, FL (NOAA Fisheries 2009b). Using data from Caldwell's (2001) study, Gubbins et al. (2003) calculated the dolphin abundance within the inshore and coastal waters of Jacksonville to be 412 individuals. This abundance estimate is the only one available for the JES stock and is the number that is used in the current 2009 stock assessment report, yet it includes both estuarine and coastal dolphins. Thus, an updated abundance estimate for estuarine dolphins is much needed; especially considering that the JES stock is currently classified as a strategic stock due to an estimated small population size (NOAA Fisheries 2009b). In addition, the population status of

dolphins within the SJR is of particular interest due to an unusual mortality event (UME) that occurred within the river in 2010 and the potential impacts of the 2013-2014 UME along the Atlantic coast. In contrast to prior reports which stated that the southern community was comprised solely of seasonal residents with summer site fidelity (Caldwell 2001), recent preliminary analyses indicate that the SJR population consists of year round residents, seasonal residents, and transients (Gibson, unpublished data), thus increasing the risk of cumulative anthropogenic disturbance.

The aim of this study was to use a mark-recapture model approach to estimate population abundance and determine seasonal variations and rates of emigration for the SJR dolphins. This aim was accomplished using two study designs. The first approach was to model abundance and emigration utilizing a full year of data with weekly surveys. The second approach was to model the same parameters, but instead utilizing data from less frequent seasonal surveys conducted over three years. Based on previous analyses for this population, abundance was expected to be greatest in the summer and least in the winter, and the emigration rate was expected to be largest between summer and autumn. This study will provide current population estimates necessary for informed management decisions and will also enable a comparison of estimates obtained via two different sampling methods.

Methods

Study Area

The SJR is a large brackish river that spans nearly 500 km through Northeast Florida (DeMort 1991), and is unusual in that it flows from south to north. At the lower end of the SJR, it crosses the intracoastal waterway (an inland waterway that parallels the coast) and flows into

the Atlantic Ocean. The lower SJR is a metropolitan/industrial area that is characterized by extensive boat traffic and shipping activity due to the presence of an international shipping port, naval and Coast Guard stations, and commercial fishing fleets.

Field Methods

Vessel-based photo-identification surveys were conducted along a fixed survey route, following the main channel from the mouth of the river in Mayport (N30.39904, W-81.39396) to 40 linear kilometers upriver in downtown Jacksonville (N30.31479, W-81.62987), and direction of travel was alternated for each survey (Figure 1). This survey route extends well beyond that of the 1994-1997 study and the 40 km stopping point was chosen as salinity approaches zero at this location. The research vessel was either a 6.4 m Carolina skiff or an 8 m TwinVee catamaran with a minimum of three personnel onboard. The vessel was operated at a speed of 10-12 km/h until dolphins were spotted, at which point the vessel was slowed to match the speed of the dolphins or stopped completely. Using a professional grade digital SLR camera equipped with a 100-400 mm telephoto zoom lens, photographs were taken of the dorsal fins of each individual and the sighting location was recorded with a hand-held global positioning system (GPS). For each sighting, the minimum, maximum, and best field estimates were recorded for total group size as well as the number of calves and young of the year. Group membership was defined using the conservative 10 meter chain rule (Smolker et al. 1992).

Photo Analysis

Photographic data were processed using standard photo identification practices (Mazzoil et al. 2004). The best photograph of each individual from each sighting was selected. These photographs were then graded on focus, contrast, angle, proportion of fin visible, and proportion of frame filled by fin and subsequently received a rating of Q-1 (excellent quality), Q-2 (average

quality), or Q-3 (poor quality) (Urian et al. 1999). Only Q-1 and Q-2 photographs were included in analyses. All identified individuals then received a distinctiveness score of D-1 (very distinctive), D-2 (average distinctiveness), or D-3 (not distinctive) (Urian et al. 1999). Analyses were restricted to individuals that ranked as D-1 or D-2. The best photograph of each individual in each sighting was compared to the master catalog of previously identified individuals and the sighting history for each individual was updated. If no match was found, the dolphin was entered as a new individual and given a new identification code.

Mark-Recapture Analysis

One commonly used method of obtaining abundance data for cetaceans is through mark-recapture studies, which have been shown to yield relatively precise estimates of population size (Read et al. 2003; Pleslić et al. 2013). Bottlenose dolphin abundance estimates have typically been generated utilizing two general types of mark-recapture models: closed and open. Closed models are used when the population is assumed to have no births, deaths, immigration, or emigration within the sampling period, whereas open models allow for emigration, immigration, births, and/or deaths to occur within the sampling period. For populations with temporary emigration, the robust design can be incorporated to combine features of both open and closed models (Pollock 1982). Temporary emigration can be categorized as completely random or Markovian. When a population is composed of residents, seasonal residents, and transients, emigration can no longer be considered completely random (Conn et al. 2011). Markovian emigration assumes that individuals that have temporarily emigrated out of the study area are more likely to stay out of the study area during the next time period than individuals that stayed within the study area during the previous time period (Kendall et al. 1997). Thus, based on

current understanding of residency patterns in the SJR, the robust design with Markovian temporary emigration was expected to provide the best fit for these data.

For both the one year and three year approaches, sighting histories of individual dolphins, not including calves, were analyzed in the program MARK (version 7.x; White & Burnham 1999). To determine the best fitting model for this population, the robust design with random emigration, the robust design with Markovian emigration, and the robust design with Markovian emigration and survival held constant were run. As the field methods of mark-recapture were non-invasive, no model that was adjusted for a behavioral response was included (Conn et al. 2011). The Akaike's Information Criterion values corrected for sample size (AICc) of each model were compared and the robust design with Markovian temporary emigration for a semi-closed population where survival was held constant was selected (Table 1). Survival was held constant for both the one year and three year analyses in order to make survival and emigration rates identifiable (Cooch & White 2010). This was justified by stranding data obtained from the NOAA Southeast US Marine Mammal Stranding Network, which showed that the number of bottlenose dolphin strandings in Northeast Florida did not change significantly between seasons for the time frame of this study (Chi-squared test: $P=0.86$ for the full year and $P=0.24$ for the three years). Historical stranding data that has shown no effect of season on the number of stranded animals has previously been used as justification to constrain the model in mark-recapture analyses (e.g. Speakman et al. 2010).

The one year analysis utilized weekly photo-identification survey data from June 2011 through May 2012. Sighting history data was divided into four seasons: Summer (June-August), Autumn (September-November), Winter (December-February), and Spring (March-May). Over the 12 month period, a total of 48 surveys were conducted. Within the robust design, there were

4 primary occasions: summer, autumn, winter, and spring, and 48 secondary occasions (Table 2a). Each secondary occasion represents one survey on a single day, approximately a week apart.

For the three-year seasonal analysis, data from August 2011 through February 2014 were used. However, these seasonal data consisted of only three surveys per summer and three surveys per winter, with each survey conducted approximately two weeks apart. Thus, the sighting history data were divided into two seasons: Summer (June-August) and Winter (January-March). For this study design, the winter season was defined by water temperatures below 18 °C; only one survey was conducted in March (March 7, 2013). During the three-year time frame, a total of 18 surveys were conducted. There were 6 primary occasions: summer 2011, winter 2012, summer 2012, winter 2013, summer 2013, and winter 2014, and 18 secondary occasions within the robust design (Table 2b). Each secondary occasion represents one survey on a single day, approximately two weeks apart. For both analyses, the population was assumed to be closed within the secondary periods, but allowed for Markovian temporary emigration between the primary periods. MARK then generated abundance (n), the temporary emigration rate for animals absent in the previous period (γ'), the temporary emigration rate for animals present in the previous period (γ''), capture probability (c), and recapture probability (p).

Results

Full Year Approach

A total of 288 individuals (non-calves) were identified between June 2011 and May 2012. The number of individuals classified as D-1 was 148, 113 as D-2, and 27 received a rating of D-3 and were excluded. The generated discovery curve shows that the identification of new

individuals appears to have leveled off with a total of 261 distinct individuals, not including calves, currently identified (Figure 2). The percentage of identified individuals that were considered distinctive was consistent across season (summer 90%, autumn 91%, winter 91%, and spring 93%). For the entire year, 31% of sightings included individuals that could not be identified, mainly due to poor photo quality. For those sightings, on average 21% of individuals in a sighting were unidentified. The percentage of unidentified individuals was greatest in autumn (26%) and least in summer (18%) and spring (18%).

The total abundance estimate for bottlenose dolphins in the SJR study area ranged from 160 to 250 individuals. The estimate for n was 250 (95% CI=243-263) in summer, 188 (95% CI=178-204) in autumn, 160 (95% CI=151-177) in winter, and 220 (95% CI=211-236) in spring (Figure 3). The γ'' estimates for the time intervals between the primary occasions were highest between summer and autumn and lowest between winter and spring (Table 3). The γ' estimates for the time intervals between the primary occasions were higher between autumn and winter than between winter and spring (Table 3). This approach yielded a capture and recapture probability ranging from 0.01 to 0.32 with a mean of 0.17.

Three-Year Seasonal Approach

A total of 218 individuals were identified between August 2011 and February 2014. Of these, 116 individuals were classified as D-1, 84 as D-2, and 18 received a rating of D-3 and were excluded. For this study period, 37% of sightings included individuals that could not be identified, mainly due to poor photo quality. Of those 37% of sightings, on average 28% of individuals in a sighting were unidentified. The discovery curve for this time period shows that the identification of new individuals appears to be steadily increasing, with a total of 200 distinctive individuals, not including calves, currently identified (Figure 4).

For the three year study period, the total abundance estimate for bottlenose dolphins in the SJR study area ranged from 67 to 210 individuals. The estimate for n was 210 (95% CI=154-316) in summer 2011, 96 (95% CI=74-148) in winter 2012, 166 (95% CI=143-204) in summer 2012, 67 (95% CI=60-87) in winter 2013, 178 (95% CI=151-222) in summer 2013, and 69 (95% CI=50-123) in winter 2014 (Figure 5). The γ'' estimates for the time intervals between the primary occasions were highest between summer and winter and lowest between winter and summer (Table 4). The γ' estimates for the time intervals between the primary occasions were highest between summer 2012 and winter 2013, and had a mean value of 0.26 (Table 4). This approach yielded a capture and recapture probability ranging from 0.00 to 0.74 with a mean of 0.26.

Discussion

These estimates provide the first assessment of bottlenose dolphin abundance within the SJR in over 16 years, and these valuable data can be utilized to generate more up to date PBR levels and an updated stock assessment. This study is also the first report of dolphins inhabiting the SJR year-round, which is an important factor in managing potential cumulative environmental impacts on this population.

Full Year Approach

Abundance Estimates

Based on the full year approach, the SJR portion of the JES was comprised of an estimated 250 individuals during the peak summer season and reached a low of 160 individuals during winter. The previous estimate of 412 individuals for the JES and surrounding coastal waters was based on data collected from late spring through autumn (May-October) and thus

reflects the peak abundance during 1997 (Gubbins et al. 2003). The new 2011 summer estimate of 250 individuals in the SJR alone is more than half the previously reported estimate for Jacksonville inshore and coastal waters. Since the SJR is just one portion of the prior study area, these findings indicate that abundance in the JES may be much greater than previously thought. In addition, it may be valuable for management agencies to assess abundance separately for each community; some dolphins within the SJR exhibit year-round residency and given the urban, industrial nature of their habitat are likely at greater risk of anthropogenic disturbance than other communities.

Transition Rates

A strong seasonal fluctuation in abundance was observed and may be a result of individuals moving into the study area during the summer mating and calving season followed by them leaving the area at the end of the season (Smith et al. 2013). The highest probability of transition to an unobservable state, given that they were present in the previous period, was estimated to occur between summer and autumn, when transients and seasonal residents emigrate out of the study area (Gibson, unpublished data). The probability of transitioning to an unobservable state was lowest between winter and spring, as the majority of individuals present in winter are year-round residents.

Three-Year Seasonal Approach

Abundance Estimates

Similar to the full year approach, abundance estimates were highest in summer and lowest in winter; this pattern was consistent across all three years. The highest abundance was estimated in summer 2011 and the lowest abundance was in winter 2013. The first year of data, summer 2011 and winter 2012, produced the highest abundance estimates for both summer and winter compared to the other two years. This inter-annual variation in dolphin abundance may

be correlated with environmental variables (e.g. prey availability and sea surface temperature); winter abundance estimates were largest in the winter with the highest average sea surface temperature (winter 2012; 17.31 °C, winter 2013: 16.33 °C, winter 2014: 14.11 °C). However, average sea surface temperatures during summer surveys did not vary substantially between years (minimum: 28.15°C, maximum: 28.66°C). Clarifying the relationship between dolphin abundance and environmental factors requires further analyses.

Transition Rates

As observed in the full year approach, the abundance of this community was consistently greatest in the summer and least in the winter. Therefore, it was expected that emigration rates would be large between summer and winter in the three-year approach. The probability of transitioning to an unobservable state, given that they were present during the previous period, was consistently greatest between summer and winter and least between winter and summer across the three year study period. The transition rate between winter 2012 and summer 2012 and between winter 2013 and summer 2013 was essentially zero, which corresponds to the seasonal shifts shown above in the full year approach. Individuals are leaving the study area after the summer mating and calving season has ended. Therefore, emigration rates are low between winter and summer as the majority of the individuals present in winter are year-round residents.

Seasonal Distribution

Given the consistent seasonal fluctuations in abundance, the question is raised as to where these estuarine dolphins go when they leave the SJR. The estuarine system approximately 170 km south of the SJR is the Indian River Lagoon (IRL). In the IRL, Durden et al. (2011) observed a seasonal fluctuation with greatest dolphin abundance in winter and least in summer, opposite the pattern for the SJR, which was proposed to be an effect of movements of the IRL

dolphins between adjacent waterways. There may be a seasonal shift in which individuals spend the summer in the SJR and head south to the northern portion of the IRL (or areas in between) during the winter; analyses are currently underway to determine the rate of interchange between these regions. Evidence of substantial intermixing may require the revision of management strategies as the populations in the SJR and IRL are currently managed as two separate estuarine stocks.

Comparing Methods

In addition to providing updated abundance estimates, this study also enables comparison of two different approaches for analyzing population size. Given that resources for research are limited, long-term studies with frequent and consistent data collection are often not feasible. Thus, many research groups rely on short bursts of intensive surveys conducted across several years (e.g. Cañadas & Hammond 2008; Dick & Hines 2011; Mansur et al. 2012). However, as displayed by the differences in discovery curves for our two study designs, limiting analyses to seasonal surveys, even with additional years, did not allow for the majority of the population to be identified. The discovery curve for the full year approach did level off, which suggests that weekly surveys provided a relatively accurate assessment of population abundance. Comparison of summer 2011 and winter 2012 abundance estimates from the two approaches indicates that the seasonal survey approach greatly underestimated the population abundance. In order to address this limitation, the seasonal survey approach could be improved by conducting additional surveys per season. Another limitation of the three-year seasonal approach was its inability to separate out seasonal shifts as clearly as the full year weekly approach. Therefore, the transition rates are not as informative for the three-year approach as they are for the full year approach. In order to elucidate these transition patterns, researchers could conduct limited surveys in the

middle of all four seasons, rather than only two. Even though there are drawbacks to limited seasonal surveys, having multiple years to compare is beneficial for assessing population stability over time.

Conservation Implications

Contrary to what has been previously reported for this population, a minimum abundance of over 150 individuals in all four seasons indicates that the SJR is an important year-round habitat for dolphins. Thus, these data suggest that population dynamics may have changed since the previous photo-identification effort in this area. With year-round utilization, dolphins are more susceptible to cumulative effects of anthropogenic disturbances. This study has provided baseline data onto which subsequent replications can be added and abundance estimates should be reassessed as more data becomes available. Such data will play an important role in determining when environmental changes, either natural or anthropogenic, may be affecting this population's health or movement patterns.

As data on the SJR community was outdated prior to the UME in 2010, the cause of the die-off remains undetermined, but dredging operations in the river were considered as a potential contributing factor (Jackson 2010). The U.S. Army Corps of Engineers has recently approved a large-scale harbor expansion project which will deepen the main channel of the St. Johns River by over 2 m from the mouth to 21 km upriver (Bostick 2014), and is intended to increase shipping traffic. Research has shown that extensive dredging operations have resulted in the displacement of dolphins from the area, even when the dolphin population was presumably already habituated to maintenance dredging and shipping activities (Pirotta et al. 2013). We hope that the findings presented here, which suggest that the estuarine waters of the SJR provide

significant habitat for bottlenose dolphins, will enable management agencies to better assess the conservation issues associated with the proposed SJR harbor expansion project.

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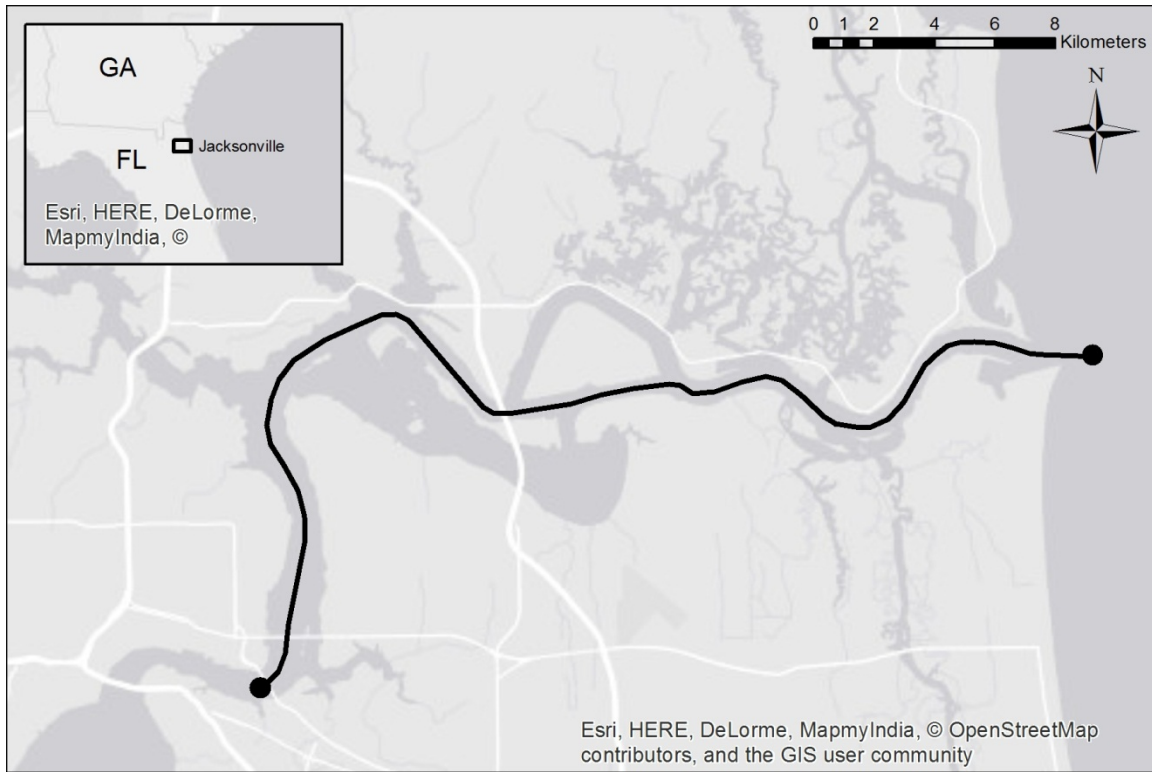


Figure 1. Map of the Jacksonville, Florida, USA study area. The survey route followed the main channel of the river and is designated by the line. Start/end locations are designated with dots; direction of travel was alternated each week.

Table 1. Model statistics for mark-recapture models fitted to capture histories of bottlenose dolphins in the St. Johns River, FL. The full year weekly survey approach above and the three-year seasonal survey approach below.

FULL YEAR						
Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
$\varphi_{(t)}\gamma''_{(t)}=\gamma'_{(t)} p_{(t)}\neq c_{(t)}$	2681.8846	0.0000	1.00000	1.0000	97	5891.3069
$\varphi_{(t)}\gamma''_{(t)}\neq\gamma'_{(t)} p_{(t)}=c_{(t)}$	2772.3597	90.4751	0.00000	0.0000	58	6067.7303
$\varphi_{(.)}\gamma''_{(t)}\neq\gamma'_{(t)} p_{(t)}=c_{(t)}$	2773.3563	91.4717	0.00000	0.0000	58	6068.7268
$\varphi_{(t)}\gamma''_{(t)}\neq\gamma'_{(t)} p_{(.)}=c_{(.)}$	3079.7885	397.9039	0.00000	0.0000	11	6473.2996
$\varphi_{(.)}\gamma''_{(t)}\neq\gamma'_{(t)} p_{(.)}=c_{(.)}$	3080.1332	398.2486	0.00000	0.0000	11	6473.6443

THREE YEAR						
Model	AICc	Delta AICc	AICc Weight	Model Likelihood	No. Par.	Deviance
$\varphi_{(t)}\gamma''_{(t)}=\gamma'_{(t)} p_{(t)}\neq c_{(t)}$	-766.7179	0.0000	0.99859	1.0000	44	335.3871
$\varphi_{(t)}\gamma''_{(t)}\neq\gamma'_{(t)} p_{(t)}=c_{(t)}$	-753.5536	13.1643	0.00138	0.0014	34	371.6397
$\varphi_{(.)}\gamma''_{(t)}\neq\gamma'_{(t)} p_{(t)}=c_{(t)}$	-745.4513	21.2666	0.00002	0.0000	38	370.6108
$\varphi_{(t)}\gamma''_{(t)}\neq\gamma'_{(t)} p_{(.)}=c_{(.)}$	-612.0633	154.6546	0.00000	0.0000	17	550.4635
$\varphi_{(.)}\gamma''_{(t)}\neq\gamma'_{(t)} p_{(.)}=c_{(.)}$	-603.8329	162.8850	0.00000	0.0000	21	550.1176

The Robust design with full likelihood probability of capture and recapture was selected for all models. Akaike information criterion corrected for small sample size (AICc) was used to select the best fitting model.

φ survival

γ'' & γ' temporary emigration

p probability of capture

c probability of recapture

$.$ represents the parameter being held constant

t represents the parameter varying over time

$\gamma''=\gamma'$ denotes random emigration

$\gamma''(t)\neq\gamma'(t)$ denotes Markovian emigration

Table 2. Sampling design: Number of surveys for each time period, separated into primary and secondary occasions. The full year weekly survey approach above (1a) and the three-year seasonal survey approach below (1b).

Full year design						
Year	2011		2012			
Primary Occasion (Season)	Summer	Autumn	Winter	Spring		
Secondary Occasion (No. of surveys)	14	12	10	12		

Three-year design						
Year	2011	2012		2013		2014
Primary Occasion (Season)	Summer	Winter	Summer	Winter	Summer	Winter
Secondary Occasion (No. of surveys)	3	3	3	3	3	3

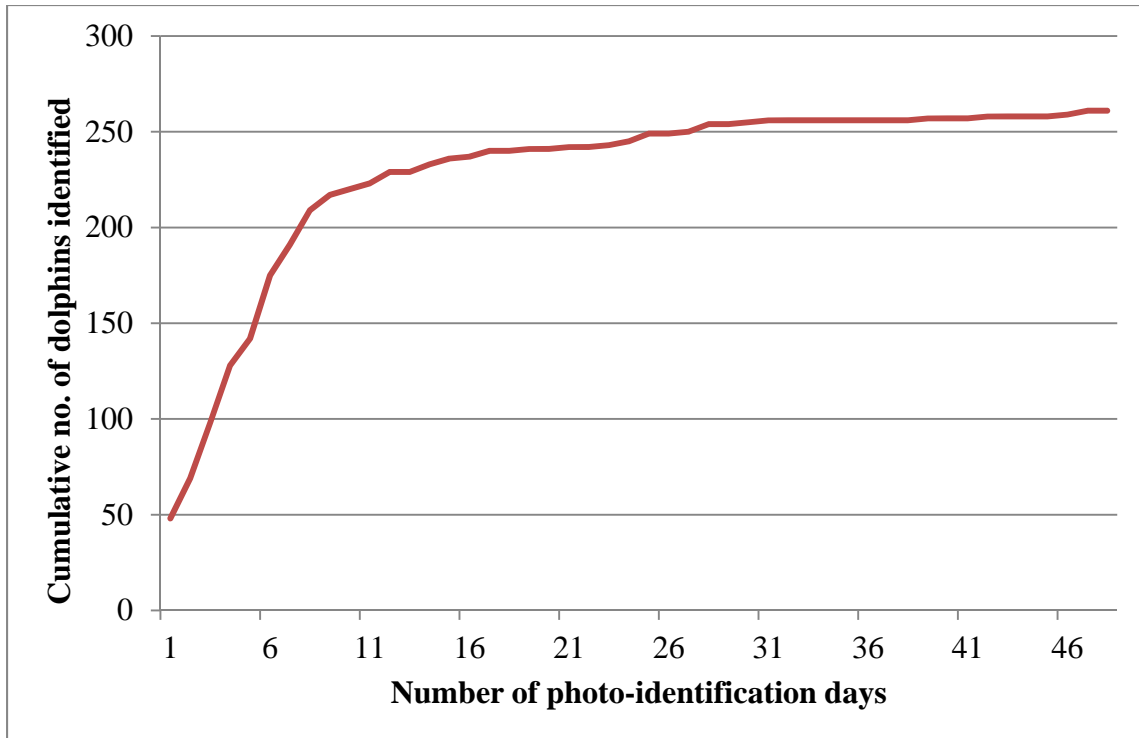


Figure 2. Discovery curve of newly identified dolphins, not including calves, during a full year of weekly surveys from June 2011-May 2012. Data were restricted to individuals ranking as D-1 or D-2 in terms of distinctiveness.

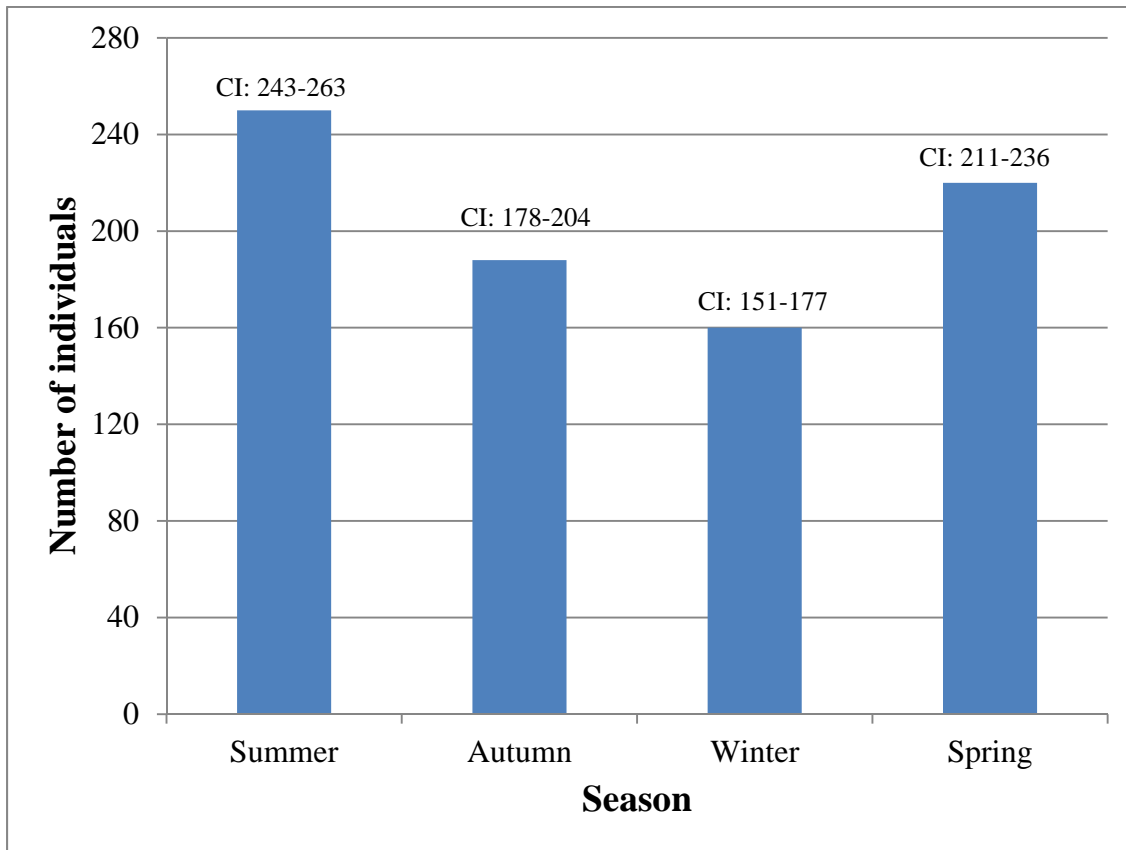


Figure 3. Seasonal abundance estimates of individuals, not including calves, for the full year approach from June 2011 to May 2012. The 95% confidence intervals are presented above each bar.

Table 3. Temporary emigration estimates for animals present in the previous period (γ'') and the temporary emigration rate for animals absent in the previous period (γ') for the period between each primary sampling occasion of the full year approach from June 2011 to May 2012.

Period of Interchange	Temporary Emigration Rates			
	(γ'')	95% Confidence Interval	(γ')	95% Confidence Interval
Summer – Autumn	0.29	0.23 – 0.35	-	-
Autumn – Winter	0.15	0.08 – 0.26	1.00	0.99 – 1.00
Winter – Spring	0.04	0.01 – 0.20	0.36	0.25 – 0.49

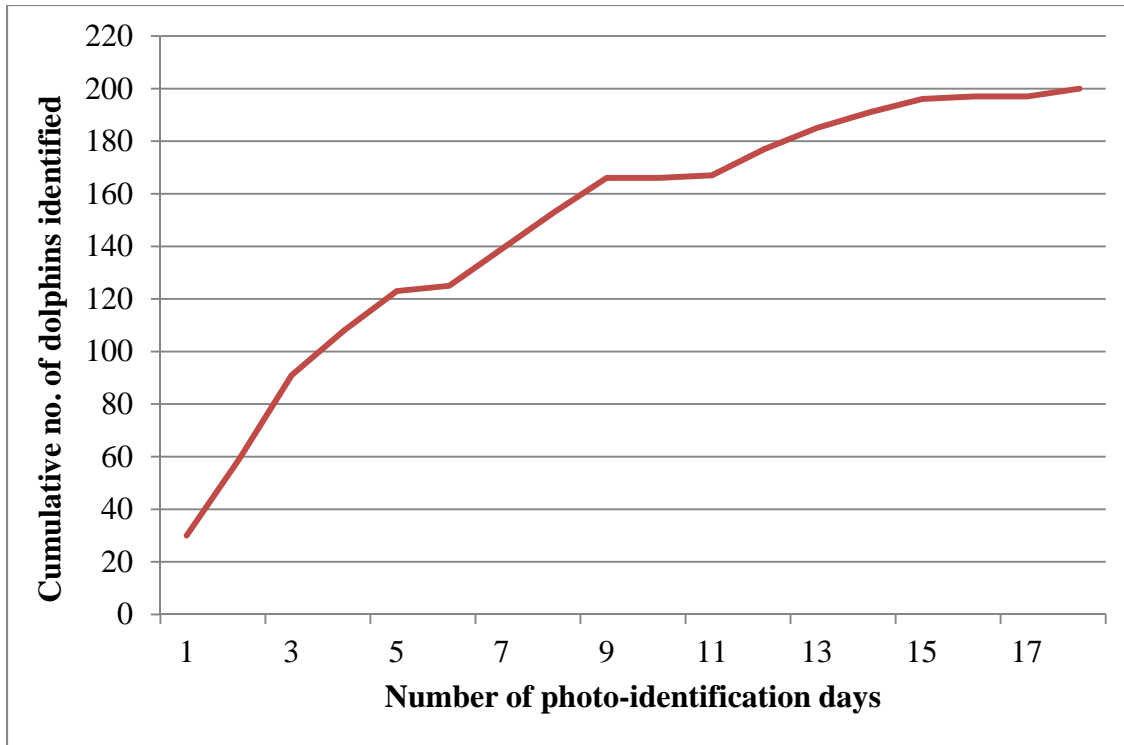


Figure 4. Discovery curve of newly identified dolphins, not including calves, during the three years of seasonal surveys from August 2011 and February 2014. Data were restricted to individuals ranking as D-1 or D-2 in terms of distinctiveness.

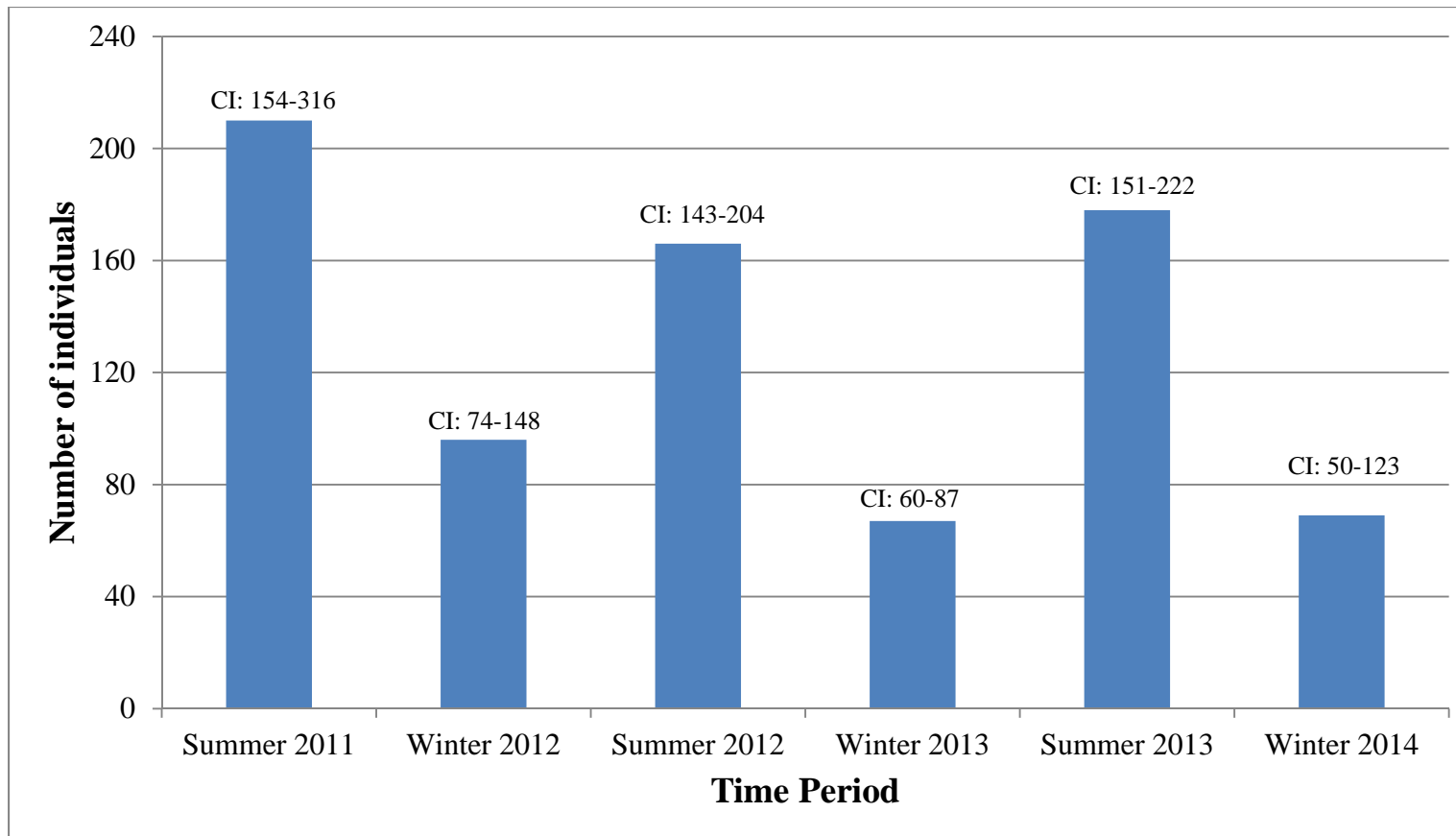


Figure 5. Seasonal abundance estimates of individuals, not including calves, for the three-year approach from August 2011 to February 2014. The 95% confidence intervals are presented above each bar.

Table 4. Temporary emigration estimates for animals present in the previous period (γ'') and the temporary emigration rate for animals absent in the previous period (γ') for the period between each primary sampling occasion of the three-year approach from August 2011 to February 2014.

Period of Interchange	Temporary Emigration Rates			
	(γ'')	95% Confidence Interval	(γ')	95% Confidence Interval
Summer 2011 – Winter 2012	0.55	0.33 – 0.74	-	-
Winter 2012 – Summer 2012	0.00	0.00 – 0.00	0.36	0.12 – 0.70
Summer 2012 – Winter 2013	0.65	0.53 – 0.75	0.59	0.19 – 0.90
Winter 2013 – Summer 2013	0.00	0.00 – 0.00	0.10	0.02 – 0.40
Summer 2013 – Winter 2014	0.66	0.44 – 0.82	0.00	0.00 – 0.00

Chapter 2

The Effects of Study Area Size on Home Range Estimates of Bottlenose Dolphins (*Tursiops truncatus*)

Abstract

Knowledge of an animal's home range is a crucial component in making informed management decisions. However, many home range studies are limited by the study area size, and therefore may underestimate the size of the home range. In many cases, individuals have been shown to travel outside of the study area and utilize a larger area than estimated by the study design. In this study, data collected by multiple research groups studying bottlenose dolphins on the east coast of Florida were combined to determine how home range estimates increased with increasing study area size. Home range analyses utilized photo-identification data collected in the St. Johns River (SJR; Jacksonville, FL) and adjacent waterways, extending a total of 253 km to the southern end of Mosquito Lagoon in the Indian River Lagoon Estuarine System. Univariate kernel density estimates (KDE) were computed for individuals with 10 or more sightings (n=20). Sighting coordinates were projected onto a line that ran down the middle of all study areas and distances from the origin were measured for each point. As each study area was not surveyed with a uniform effort, sightings from each area were weighted.

Kernels were calculated for the primary study area (SJR) first, then additional kernels were calculated by combining the SJR and the next adjacent waterway; this continued in an additive fashion until all study areas were included. The 95% and 50% KDEs calculated for the SJR study area alone ranged from 21 to 35 km and 4 to 19 km, respectively. The 95% and 50% KDEs calculated for all combined study areas ranged from 116 to 217 km and 9 to 70 km, respectively. This study illustrates the degree to which home range may be underestimated by the use of limited study areas and demonstrates the benefits of conducting collaborative science.

Introduction

The home range of an animal is defined as the area occupied by an individual during its everyday activities (Burt 1943). The area in which an animal spends 95% percent of its time is considered the home range, while the more concentrated area in which it spends 50% of its time is the core area (White & Garrot 1990). In terrestrial mammals, it has been shown that home ranges are generally larger for carnivores than for herbivores (Harestad & Bunnell 1979). In addition, home range sizes have been linked to the body size of an animal (McNab 1963; Garland 1983; Lindstedt et al. 1986). As bottlenose dolphins are one of the top marine predators and capable of swimming very efficiently, they are expected to have large home ranges (Ingram & Rogan 2002). Male bottlenose dolphins can weigh up to 500 kg and females up to 260 kg (Folkens et al. 2008). Terrestrial mammals of this size have a predicted home range of 396.03 km² for males and 214.18 km² for females (Lindstedt et al. 1986). Although locomotion constraints are typically limiting for terrestrial mammals, in an aquatic environment this is no longer the case; therefore, bottlenose dolphin home ranges can be larger for the same level of energetic costs (Williams et al. 1992; Connor et al. 2000). Male bottlenose dolphins presumably

have larger home ranges than females, which are thought to allow increased reproductive access to females (Eisenberg 1966; Wells et al. 1987; Wells 1991).

Home range studies are typically conducted to provide data on the extent and area of habitat use in order to make spatial planning and management decisions, and potentially lead to the evaluation of anthropogenic impacts on populations (Merriman et al. 2009). However, one substantial limitation of such studies is that the area estimated to be the home range is often limited by the size of the study area (Zolman 2002; Merriman et al. 2009; Urian et al. 2009). Many studies have used only the limited data from their study area to quantify home ranges, even though they have documented individuals travelling far beyond the boundaries of the study area (Gruber 1981; Hanby 2005; Fury & Harrison 2008). For example, Ingram & Rogan (2002) concluded from their study of bottlenose dolphins in the Shannon Estuary in Ireland, that the study estuary did not encompass the entire geographical range of the population because the ranges of individuals did not reach an asymptote when all of their sightings were included. Similarly, it appears that a chosen study area in the Marlborough Sounds, New Zealand is only one important section of a larger home range as dolphins have been estimated to travel more than 80 km outside of the study area (Merriman et al. 2009). Thus, it is unlikely that the boundaries of a selected study area will encompass the entire home range of an individual dolphin (Davis 1953), as the animals are not constrained by the study area boundaries.

Data on bottlenose dolphin home ranges have been based on study areas varying from 100-12,000 km², resulting in home range estimates ranging from 32-343.89 km², and the percent coverage (home range/study area x 100) ranging from 1.08-104% (Shane 1987; Connor et al. 2000; Gubbins 2002; Ingram & Rogan 2002; Lynn & Wursig 2002; Candido & Dos Santos 2005; Hanby 2005; Litz et al. 2007; Martinez-Serrano et al. 2011; Kiszka et al. 2012). In terms

of linear distances, bottlenose dolphins have been reported to have a minimum range from 50 – 470 km (Defran et al. 1999). Thus, there is great variability in home range estimates currently available for bottlenose dolphins, which could be due to the fact that funding and time often constrain the area a single research team can cover. Extending the boundaries of individual study areas into adjacent waterways, thereby covering more of an animal's range, would enable better conservation of this species through improved knowledge of their movements (Ingram & Rogan 2002).

While previous research suggests that the size of the study area impacts the estimate of home range, to our knowledge no study has directly tested for a change in estimated home range size with increased study area. The Northeast Florida Dolphin Research Consortium, a collaboration among eight research organizations with adjacent estuarine study areas, provides a unique opportunity to assess the impact of study area size on home range and core area estimates. Within the area covered by the consortium, there are currently two recognized estuarine stocks of bottlenose dolphins: the Jacksonville Estuarine System stock and the Indian River Lagoon Estuarine System stock. These two stocks are currently managed separately as rates of interchange are thought to be low. This research compares the home range and core area estimates from a 40 linear km study area within the St. Johns River (SJR; Jacksonville, FL) with the home range and core area estimates obtained by sequentially adding adjacent study areas, and ultimately including the Consortium's combined 253 km study area.

Methods

Study Area

The SJR served as the primary study area for this project. It spans nearly 500 km through Northeast Florida (DeMort 1991) and is a large brackish river with extensive boat traffic and shipping activity. The SJR intersects the intracoastal waterway (ICW; an inland waterway that runs north to south paralleling the coast) 8 km from the mouth of the river. Approximately 170 km to the south, the ICW converges with the Indian River Lagoon system. The individual study areas for these analyses consist of the SJR (40 km), the ICW south through Duval county (DUV; 55 km), the ICW south through St. John's county (SJC; 36 km), the ICW south through Flagler county (FL; 31 km), the ICW south to New Smyrna Beach (NSB; 48 km), and lastly, south through the Mosquito Lagoon of the Indian River Lagoon Estuarine System (ML; 43 km) (Table 1; Figure 1).

Data Collection

The Northeast Florida Dolphin Research Consortium was established in 2011 to coordinate research efforts in response to a bottlenose dolphin Unusual Mortality Event in the St. Johns River. The initial goal of the consortium was to systematically survey bottlenose dolphins in Northeast Florida's estuarine waters to determine abundance and rates of inter-change between regions. During seasonal coordinated surveys, each organization was responsible for surveying one section of the 253 km study area.

Each survey was conducted along a fixed survey route by a team of at least three personnel consisting of a boat driver, photographer, and data recorder. The vessel was operated at a speed of 10-12 km/h until dolphins were spotted, at which point the vessel was slowed to match the speed of the dolphins or stopped completely. Using a professional grade digital

camera equipped with a 100-400 mm telephoto zoom lens, photographs were taken of the dorsal fins of each individual and the sighting location was recorded with a hand-held global positioning system (GPS). For each sighting, the minimum, maximum, and best field estimates were recorded for group size as well as the number of calves and young of the year.

Surveys were conducted from March 2011 through May 2012, followed by an additional 12 surveys from April 2012 through February 2014 for all areas except SJC and FL. The frequency of surveys differed for each study area. The SJR was surveyed on a weekly basis, with approximately four surveys a month. In DUV, SJC, and FL, surveys were conducted opportunistically, with a maximum of four surveys per season in DUV and three in SJC and FL. NSB was surveyed approximately twice monthly and in ML there were approximately 3 surveys every summer and 3 surveys every winter. The total number of surveys conducted in each section was as follows: SJR (71), DUV (23), SJC (17), FL (18), NSB (41), and ML (19). Due to the variation in survey frequencies, analyses of home range were weighted based on effort in each area.

Photo Analysis

Individual dolphins were identified using standard photo identification practices (Mazzoil et al. 2004). The best photograph of each individual from each sighting was selected and graded for quality. Quality was defined by focus, contrast, proportion of fin visible, proportion of frame filled by fin, and angle (Urian et al. 1999). Each photograph received a score of Q-1 (excellent quality), Q-2 (average quality), or Q-3 (poor quality). Only Q-1 and Q-2 photographs were included in analyses. All individuals then received a distinctiveness score of D-1 (very distinctive), D-2 (average distinctiveness), or D-3 (not distinctive) (Urian et al. 1999). Only individuals ranked as D-1 or D-2 were included in analyses. The photographs were then

compared to the master catalog for the relevant study area and the sighting history for each individual was updated. If no match was found, the dolphin was entered as a new individual and given a new identification code. Photographs from the main (SJR) catalog, excluding calves, were then matched against photographs provided by the other consortium organizations. For matched individuals, all sighting information from March 2011 – February 2014 was obtained and consolidated.

Range Calculations

Individuals from the main catalog that were also sighted outside of the SJR study area were selected (n=27); 18 ranked as D-1, nine as D-2, and none were ranked as D-3. It is important to note that the number of individuals sighted outside of the SJR is a conservative number as the entire catalogue of each organization was not exhausted; thus, these data do not reflect the rates of interchange between these regions. Of the selected individuals, only dolphins that were sighted 10 or more times were included in the analyses (n=20).

Sighting histories of individuals with GPS location data were plotted in ArcGIS 10.1 (ESRI; Redlands, California). A midline was mapped throughout the entire geographic area covered, and the GPS locations were transformed onto the line using the “locate features along routes” function. The furthest upriver location that was surveyed in the SJR was defined as location zero. Distances from location zero were then computed for each of the sighting locations on the line, resulting in a univariate data set. Due to the fact that the end of the SJR does not lead into the next study area, but instead ends at the ocean, the midline through the SJR was truncated at the intersection with the ICW and continues south to the DUV study area (Figure 2). The intersection lies approximately 8 km west of the mouth of the river. Therefore, sightings east of the intersection were condensed onto the nearest point of the shortened midline.

The decision was made to place location zero on the west side of the ICW intersection rather than at the mouth of the river because 80% of the sightings in the SJR were located on the west side. Therefore, more sightings would have been condensed if location zero was set to the east side of the intersection. In order to assess the potential effects of this truncation; home range analyses were conducted for the SJR study area using both the shortened midline and a midline extending the full length of the study area; differences between the two estimates were then calculated.

Maximum Linear Distance

The maximum linear distance was calculated to determine the distance between the two most extreme sightings of each individual, without crossing land. Using the univariate data set produced in ArcGIS, maximum linear distances were first calculated for individuals sighted 10 or more times in the SJR study area. The consortium data was then added following a sequential order from the SJR study area south; DUV, SJC, FL, NSB, and ML. When the number of sightings for each individual reached 10 or above, the maximum linear distance was computed using the combined sighting location data.

Home Range Analyses

Due to the high variability in home range estimates, a number of studies have been conducted that compare the precision and accuracy of the various home range estimators. The Kernel Density Estimator (KDE) is considered the most robust (Hansteen et al. 1997), accurate, efficient (Borger et al. 2006), and beneficial test for home range analyses (Bowman & Azzalini 1997). Typically, home range studies have utilized bivariate data to estimate space use, not accounting for any barriers to the animal's movements (Vokoun 2003) until after the polygons of space use are computed, at which point the unusable area is often removed (e.g. Fury & Harrison 2008; Gibson et al. 2013). However, the removal of uninhabitable area (i.e. land) from bivariate

home range kernels is problematic because the proportion removed is not uniform across individuals. As an alternative, the use of univariate data for home range analyses is beneficial when studying a population that inhabits a narrow, aquatic environment. Spatial density calculation (Hanby 2005) is another potential method of limiting analyses to space that can be utilized by the animal, but in a narrow habitat, utilizing a second dimension in space is not necessarily more informative. Thus, the KDE was used in conjunction with univariate data as it allows for a more accurate analysis of the home ranges (Moyer et al. 2007) in a linear estuarine environment. This method of analyzing univariate data with KDE has also been used to calculate alongshore home ranges (Rayment et al. 2009).

Home ranges were first calculated for individuals sighted 10 or more times within the SJR study area. Then data from each additional study area was added following a sequential order. Once the number of sightings for each individual reached 10 or above, the home range was calculated using the combined sighting location data. Each sighting was weighted based on the survey effort in each study area (Equations 1 & 2; Rayment et al. 2009). Each study area was assigned a weight, W_i , which was calculated using Formula 1 where A_i = the area of each section that was surveyed, V_i = the number of times that section was surveyed during the chosen time period, and T = the number of sections surveyed.

$$W_i = 1 / \frac{(A_i \times V_i)}{\sum_{i=1}^T (A_i \times V_i)} \quad (1)$$

Each sighting then received a scaled weight, W_i^S , which was calculated using Formula 2 where N = the total number of sightings for each individual.

$$W_i^S = \frac{W_i}{\sum_{i=1}^N W_i} \quad (2)$$

The univariate data for each individual was input into the program SAS (version 9.2, SAS Institute, Cary, NC) and the function PROC KDE was used to compute the fixed 95% and 50% utilization KDE by incorporating the weights into the program. Three different computational methods for automatically selecting bandwidth were used: simple normal reference (SNR), Silverman's rule of thumb (SROT), and the Sheather-Jones plug-in (SJPI). The resulting kernel density graphs were then compared. With an increase in sightings, the pattern between bandwidth smoothing methods became apparent (Figure 3). SROT undersmoothed the data and showed more of a bimodal distribution for many of the individuals. SNR and SJPI produced very similar graphs; however, SJPI oversmoothed and therefore obscured the underlying structure. Lastly, SNR appeared to moderately smooth the data, and therefore was used for all analyses. The home ranges for all individuals were first estimated utilizing only SJR data, then the adjacent study area's data were added to SJR's and the new home range was estimated. This continued one by one until all of the study areas were included.

Results

Sightings

Twenty-seven distinctive individuals were sighted in the SJR study area and they had an average of 8.44 sightings within the SJR (Table 2). One dolphin was sighted in six study areas, two dolphins were sighted in five study areas, six were sighted in four study areas, eight were sighted in three study areas, and ten were sighted in two study areas. However, these were not necessarily adjacent study areas. Of the individuals sighted in only two study areas, 50% of them were seen in SJR and ML, the two furthest locations in this study (Table 2).

Maximum Linear Distance

When solely looking at the 40 km SJR study area, the distance between individuals' two most extreme points ranged from 15.97 to 28.43 km. When increasing the study area size to 95 km, individuals' maximum linear distance ranged from 11.48-73.98 km, for the 131 km study area it was 80.05-109.31 km, for the 162 km study area it was 112.20 km, for the 210 km study area it was 171.66-190.31 km, and for the 253 km study area it was 170.05-215.96 km (Table 3). Of the 10 individuals sighted in the furthest study area (ML), only three had a maximum linear distance above 200 km. The smallest maximum linear distance was 11.48 km, which was for an individual (SLPY) sighted in the adjacent SJR and DUV study areas.

Home Range Estimates

The 95% kernel estimates ranged from 20.65-35.04 km for the 40 km study area, 10.64-86.67 km for the 95 km study area, 80.41-123.42 km for the 131 km study area, 138.98 km for the 162 km study area, 150.86-191.00 km for the 210 km study area, and 116.4-217.00 km for the 253 km study area (Table 4; Figure 4). Similar to the maximum linear distances, of the 13 individuals sighted in the furthest study area (ML), only two individuals had home ranges that were greater than 200 km. The 50% kernel estimates ranged from 4.38-19.4 km for the 40 km study area, 2.81-19.4 km for the 95 km study area, 8.76-65.71 km for the 131 km study area, 10 km for the 162 km study area, 10.00-50.92 km for the 210 km study area, and 8.76-70.09 km for the 253 km study area (Table 5; Figure 5).

The comparison of the SJR home range estimates calculated using the truncated and non-truncated midline indicated little difference between the two (Table 6). The 95% kernel density estimates from the full SJR midline ranged from 22.84-37.55 km, while for the truncated SJR midline the estimates ranged from 20.65-35.04 km. The 50% kernel density estimates from the

full midline ranged from 5.32-17.21 km, and from the truncated midline they ranged from 4.38-19.4 km.

Discussion

Maximum Linear Distance

As expected, the maximum linear distance increased as additional sightings in further study areas were added. When the full 253 km study area was used, the average maximum linear distance (190 km) falls in the middle of previously reported distances for coastal dolphins (50-470 km; Defran et al. 1999), but is much higher than mean linear distances reported for bottlenose dolphins in the Indian River Lagoon (22-54 km; Mazzoil et al. 2008). The largest maximum linear distance (215.96 km) was less than the combined 253 km combined study area, suggesting that our data may capture the true limits of these dolphins' range. In support of this, the data indicate that there may be a plateau in maximum linear distance with the inclusion of the last two study areas. These findings also demonstrate that these dolphins are not limited to one small estuarine area; some individuals traveled large distances across multiple study areas. However, one disadvantage of maximum linear distance analyses is that they do not incorporate weighting to account for differences in survey effort; they simply compute the distance from the two most extreme points. Although maximum linear distance provides valuable information on the full extent of the area travelled by an individual, it provides no information on utilization.

Home Range

Although the analyses conducted herein were for a subset of Northeast Florida's estuarine dolphins, there was a clear pattern of increasing home range estimates as study area size increased. This is an indication that an analysis consisting of the 40 km SJR study area alone

would not have properly estimated home range for these individuals given that the minimum home range was estimated at 116.40 km when all sites were included. By combining all 253 km of study area, it appears that sufficient area has been covered since the largest home range was estimated at 217 km; based on these values the percent coverage would be 86%. Home range size begins to level off when incorporating all study areas, which further supports the finding that the entire geographical range of these individuals has been reached. The findings from this study design are evidence that restricting analyses to a small study area may not give reliable information regarding the range an animal actually covers, as many of the individuals within this study inhabited more than one research group's study area. As this is one of only a few studies that have used the univariate kernel density method to date, it is not yet possible to make direct comparisons between the estimates for this study region and other areas.

In contrast to the pattern observed with home range estimates, an increase in study area size did not correspond to an increase in core area size. There was no clear pattern between the estimated core areas and the size of the study areas, suggesting that even with a small study area, researchers may be able to accurately assess core area size. Although individuals travel large distances, they appear to concentrate their time in relatively small areas. Finally, even though the analyses precluded the inclusion of a midline all the way to the mouth of the SJR, the data show that there is minimal difference in home range and core area estimates between the two methods. Thus, both the maximum linear distance and KDE estimates calculated using the truncated midline are considered to have acceptable precision.

Individual variation in ranging patterns among animals in this study was very high. For example, we had individuals that were only sighted in the SJR and ML study areas as well as individuals that used the areas in-between. The patterns suggest that individuals may be

travelling from the SJR to the southern study areas using different paths; some individuals appear to be using the ICW while others may not. The fact that individuals are not sighted in areas between the SJR and ML may indicate that these individuals are utilizing coastal areas rather than the ICW to move between regions. However, this gap of sightings between study areas may simply be due to random chance that those individuals were not in those areas on any of the survey days. Also, if those individuals are traveling through the intermediate areas quickly, they are more likely to be missed with infrequent surveys being conducted in those regions. These gaps in space affect the home range analyses in regards to how the utilization distributions are calculated, so it is important to discern whether the individuals are using these areas or not. Within the Northeast Florida region there are two large estuarine areas that have been documented providing year-round habitat for dolphins; this factor could potentially be driving the movement between the SJR and ML study areas. However, not all SJR dolphins are traveling the full distance between these estuaries.

When conducting home range studies, care must be taken beforehand to choose the appropriate method for analyses based on biological knowledge of each population and habitat. The univariate method used in this study improves upon prior studies by producing more usable information regarding the space travelled by individuals since it does not incorporate unusable area into kernel density estimates. Many studies conduct bivariate home range analyses and remove the unusable area after the estimates are produced. This method biases the estimates compared to univariate analyses. Estuarine environments tend to be linear and narrow, so this method of analyzing home range may be useful for researchers working in similar environments. Additionally, for studies in which the number of sightings for individuals are limited, it is

beneficial to use univariate data since univariate estimates require fewer sightings when compared to bivariate estimates (Vokoun 2003).

Management

In addition to addressing the fundamental research question of the effects of study area size on home range estimates, there are clear management applications for this research. A species' distribution is a key factor when making conservation management decisions, as it determines what areas need to be protected and to what extent. For example, the effectiveness of marine protected areas is likely limited by the quality of the distribution estimates used during their creation. In addition, knowledge of a population's home range and core area can provide pertinent information regarding environmental pollutants and hazards when monitoring population health (Mazzoil et al. 2008).

Bottlenose dolphin populations in the United States are managed separately as individual stocks and therefore the range of each distinct population must be known. The National Oceanic and Atmospheric Administration currently considers the dolphins in the Jacksonville Estuarine System and the Indian River Lagoon Estuarine System to be two separate populations in their management plans (NOAA Fisheries 2009b). As displayed by the large maximum linear distances and home range estimates reported here, individuals are not confined to the area currently defined as JES. Thus, individuals from these different populations may not be geographically isolated from one another. If significant mixing occurs between these two populations, then management plans may need to be revised. The Northeast Florida Dolphin Research Consortium is currently analyzing rates of interchange between the Jacksonville Estuarine System and the Indian River Lagoon Estuarine System, in an effort to address this issue. In conclusion, this study demonstrates the need to expand survey areas in order to obtain

more accurate home range estimates and thus illustrates the importance of conducting collaborative science.

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Table 1. Length of each study area along with the additive lengths of combined study areas.

	STUDY AREA					
	SJR	DUV	SJC	FL	NSB	ML
Study Area Length	40km	55km	36km	31km	48km	43km
Cumulative Length	-	95km	131km	162km	210km	253km

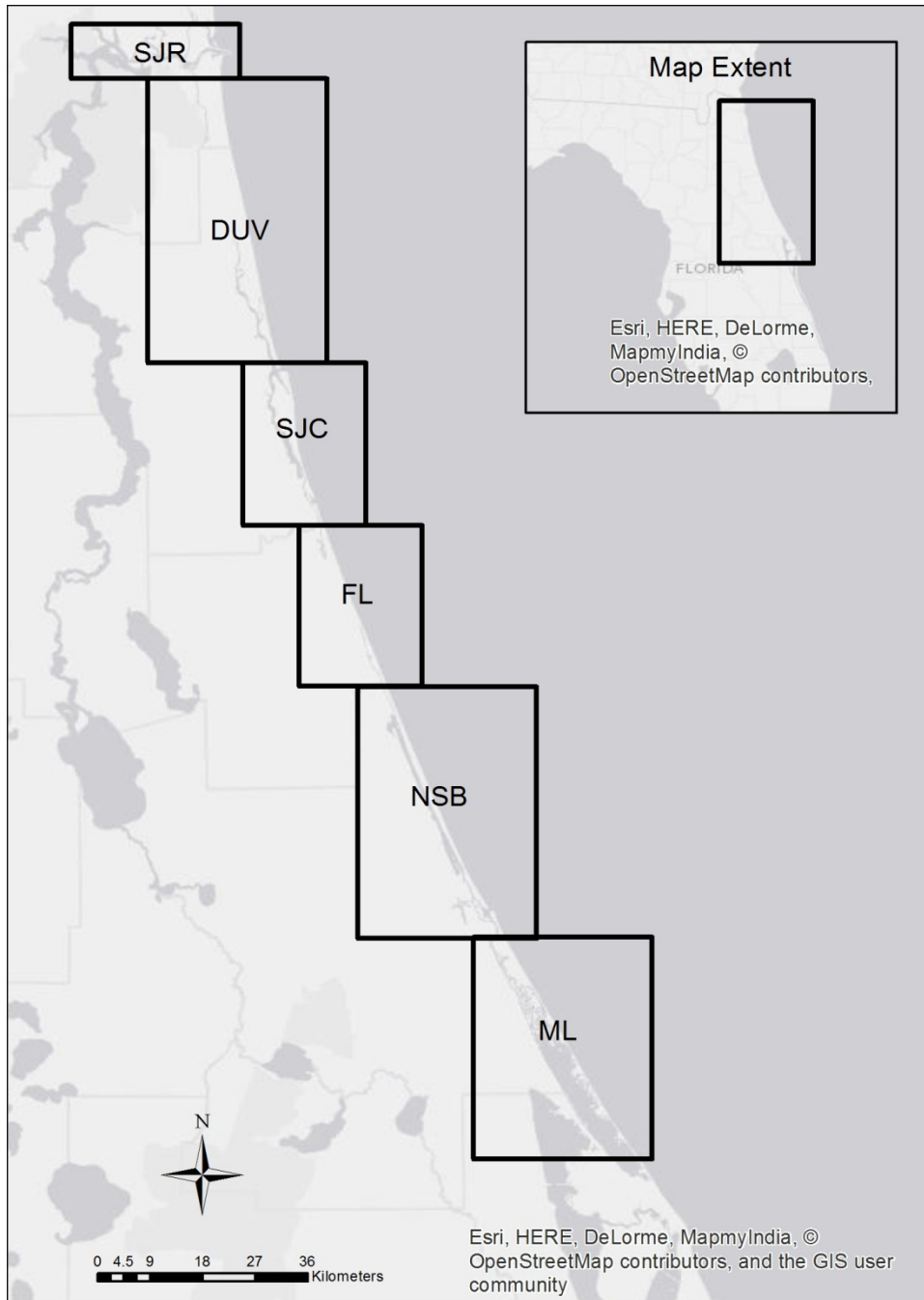


Figure 1. The six adjacent estuarine study areas: St. Johns River (SJR), the Intracoastal Waterway (ICW) south through Duval county (DUV), the ICW south through St. John’s county (SJC), the ICW south through Flagler county (FL), the ICW south to New Smyrna Beach (NSB), and south through the Mosquito Lagoon (ML).

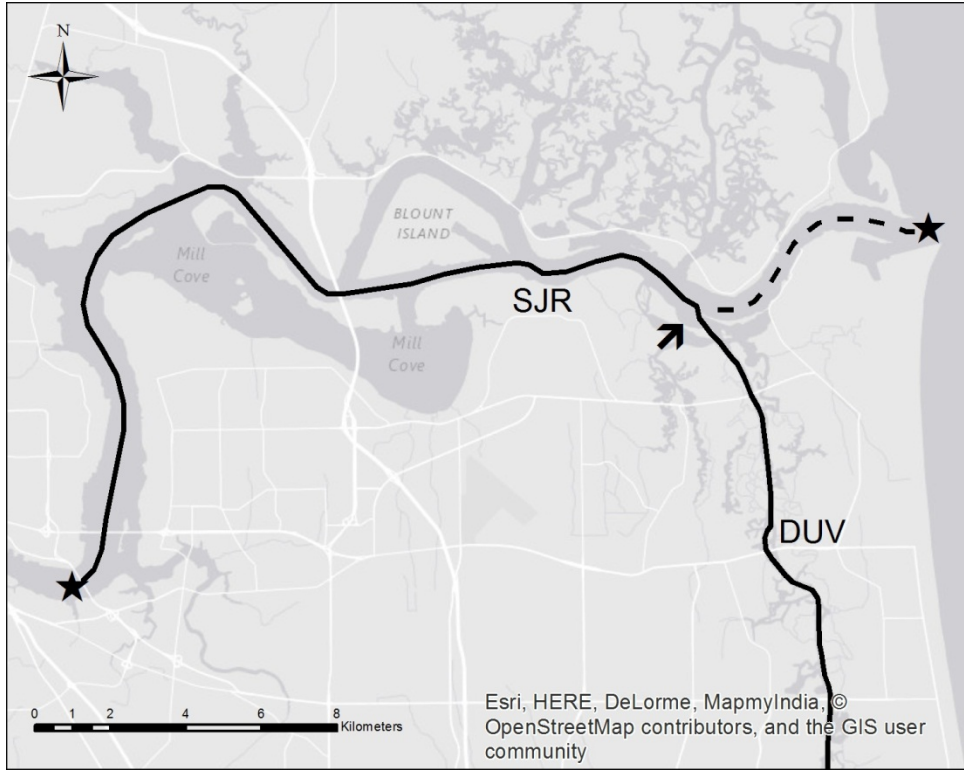


Figure 2. Map of the intersection between the St. Johns River (SJR) and the Intracoastal Waterway south through Duval county (DUV) study areas, showing the location of the truncated SJR midline. The SJR survey route is 40 km in total and 8 km (dashed line) was truncated to the intersection of the SJR and DUV study areas, displayed by the arrow. Stars represent the start and end points of the SJR survey route. The midline was chosen to begin west of the intersection versus to the east as 80% of the sightings within the SJR lied west of the intersection.

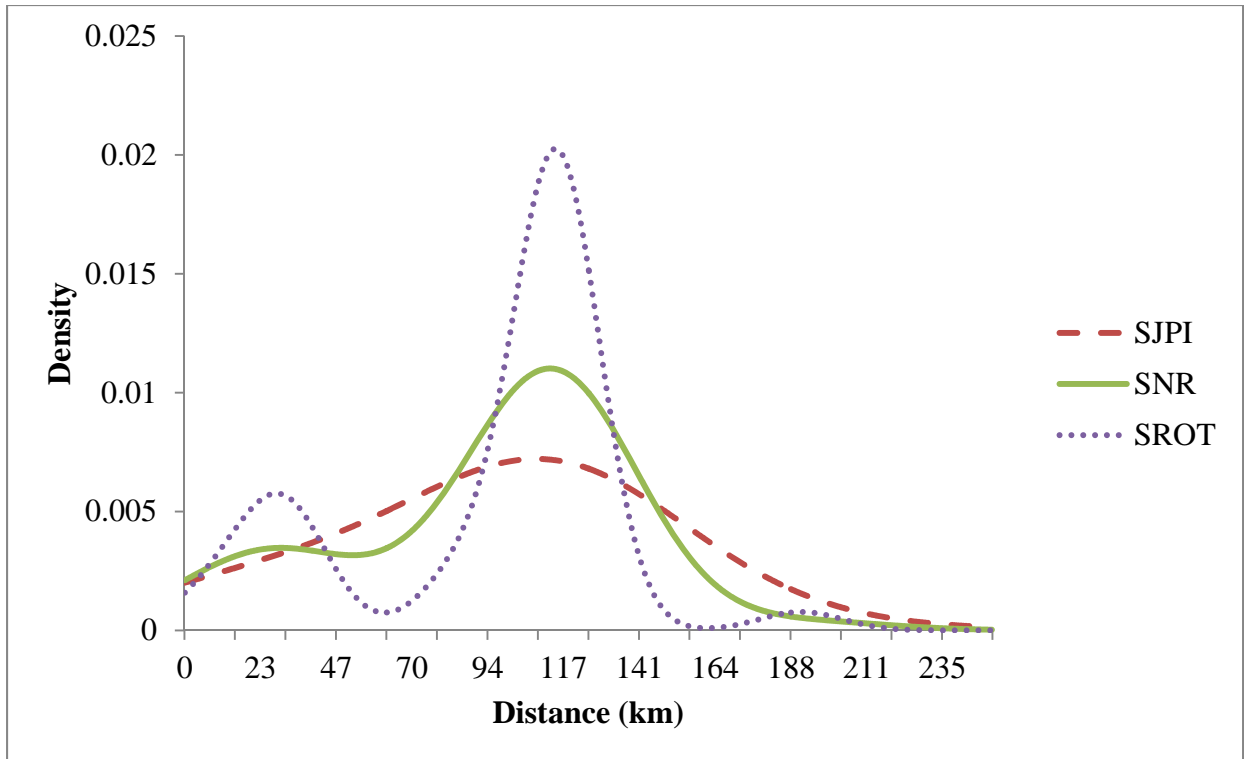


Figure 3. Univariate kernel density estimates for the distribution of sighting distances for bottlenose dolphins using the full 253 km study area. Three different bandwidth selection methods are shown: Sheather-Jones plug-in (SJPI), Simple normal reference (SNR), and Silverman's rule of thumb (SROT). SNR was selected for all analyses.

Table 2. Number of sightings for individual dolphins in each study area from March 2011 – February 2014. Blank indicates a value of zero and bold signifies ≥ 10 total sightings. Each sighting was weighted in the home range analyses based on survey effort. Animals sighted fewer than ten times were excluded from analyses.

ID CODE	STUDY AREA					
	SJR	DUV	SJC	FL	NSB	ML
NASA	16	3				
SLPY	3	11				
Q062	5	1	1			1
Q136	4	5	2			
Q139	2	5	1		1	1
Q158	8	2	1			
APLO	4	2			3	1
NUKK	6	3	2	1	1	1
Q156	1	6	3			
Q166	5	1	3			1
Q044	6	1	2			
PUKA	15				2	
Q080	10				2	
Q130	5	1		1	1	
Q122	7				2	
Q039	5	2	3		2	3
Q142	14					1
KIAW	16					1
LTUS	12	1			1	1
Q001	8					1
ZDCO	12					2
WIKD	11	1				1
Q181	7	1				1
Q144	9	1				1
NAIA	17	1				1
Q060	5					1
Q027	15	1	1			3

Table 3. The maximum linear distance (km) travelled by individuals sighted ≥ 10 times from March 2011 – February 2014, categorized by additive study areas. All animals were sighted in the 40 km study area, but maximum linear distance was not calculated until 10 or more sightings were reached.

ID CODE	STUDY AREA LENGTH (km)					
	40	95	131	162	210	253
APLO						185.10
KIAW	25.99					186.60
LTUS	19.94	21.17			171.66	182.11
NAIA	25.39	84.46				186.92
NASA	26.87	73.98				
NUKK			105.25	112.20	174.76	185.10
PUKA	26.62				190.31	
Q139						170.05
Q136			97.77			
Q142	15.97					178.36
Q166						204.97
Q144		33.69				215.96
Q156			87.85			
Q158		25.97	80.05			
Q027	28.43	29.08	109.31			190.02
Q039			99.46		178.02	184.24
Q080	19.69				172.99	
SLPY		11.48				
WIKD	27.27	33.69				215.96
ZDCO	22.51					184.69

Table 4. The 95% kernel density estimation (km) calculated for individuals sighted ≥ 10 times from March 2011 – February 2014, categorized by additive study areas. All animals were sighted in the 40 km study area but home range was not calculated until 10 or more sightings were reached.

ID CODE	STUDY AREA LENGTH (km)					
	40	95	131	162	210	253
KIAW	30.98					190
LTUS	24.09	27.85			180	190
NAIA	30.04	86.67				189
NASA	35.04	73.53				
PUKA	27.53				191	
Q142	20.65					190
Q027	34.42	35.67	109			190
Q080	24.41				181	
WIKD	34.73	43.81				217
ZDCO	27.22					190
Q158		35.99	80.41			
SLPY		10.64				
NUKK			123.42	138.98	150.86	181
Q136			85.91			
Q156			105.61			
Q139						158
APLO						191
Q039			102.85		184	194
Q166						116.4
Q144		43.81				217

Table 5. The 50% kernel density estimation (km) calculated for individuals sighted ≥ 10 times from March 2011 – February 2014, categorized by additive study areas. All animals were sighted in the 40 km study area but home range was not calculated until 10 or more sightings were reached.

ID CODE	STUDY AREA LENGTH (km)					
	40	95	131	162	210	253
KIAW	12.2					11.57
LTUS	7.51	3.44			12.52	12.52
NAIA	16.27	19.4				19.09
NASA	4.38	15.02				
PUKA	19.4				21.59	
Q142	5.01					8.76
Q027	13.14	3.13	26.28			48.19
Q080	5.94				17.83	
WIKD	9.7	10.64				20.02
ZDCO	11.26					16.27
Q158		2.81	8.76			
SLPY		2.82				
NUKK			36.95	10	10	10
Q136			16.58			
Q156			65.71			
Q139						13.45
APLO						70.09
Q039			40.6		50.92	58.43
Q166						12.52
Q144		8.45				21.59

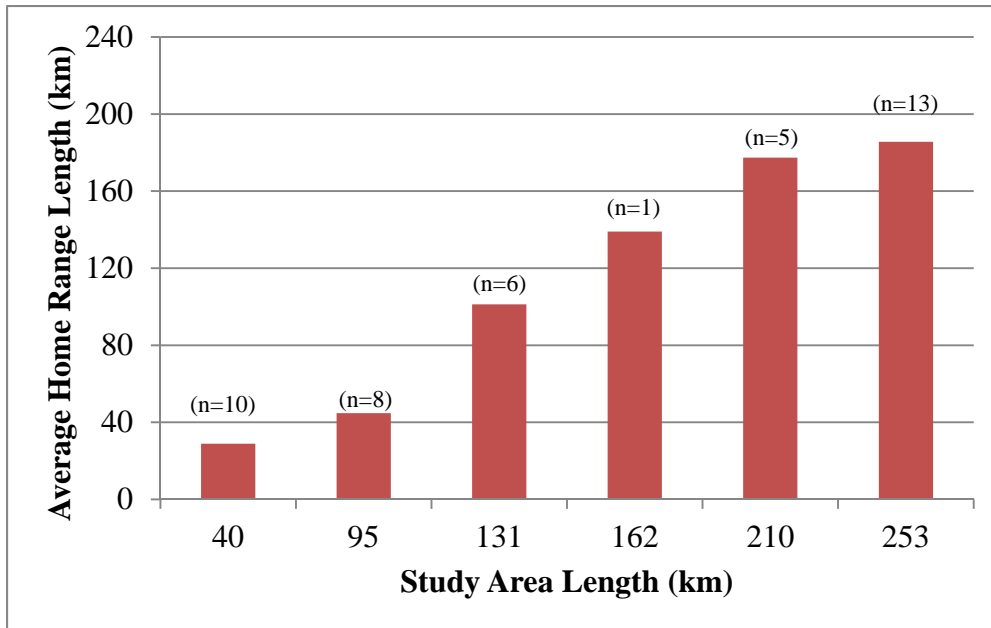


Figure 4. The average 95% kernel density estimates with increasing study area size for dolphins sighted ≥ 10 times from March 2011 – February 2014. All animals were sighted in the 40 km study area but home range was not calculated until 10 or more sightings were reached.

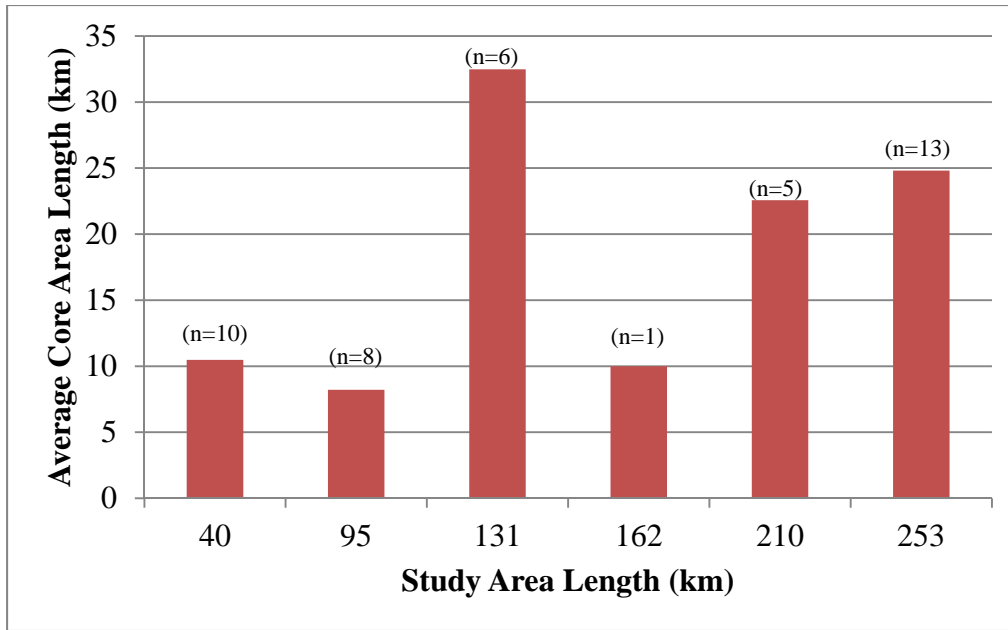


Figure 5. The average 50% kernel density estimates with increasing study area size for dolphins sighted ≥ 10 times from March 2011 – February 2014. All animals were sighted in the 40 km study area, but home range was not calculated until 10 or more sightings were reached.

Table 6. Comparison of home range and core area estimates for individuals sighted ≥ 10 times in the St. Johns River (SJR) using two versions of the midline. ‘Not truncated’ refers to the midline that extended all the way to the mouth of the SJR while ‘Truncated’ refers to the midline that did not extend the full length of the SJR study area, but instead turned south to continue on to the Intracoastal Waterway south through Duval county (DUV) study area.

ID CODE	95% HR (km)			50% CA (km)		
	Not Truncated	Truncated	Difference	Not Truncated	Truncated	Difference
KIAW	34.73	30.98	3.75	9.7	12.2	-2.5
LTUS	25.03	24.09	0.94	10.64	7.51	3.13
NAIA	30.66	30.04	0.62	16.9	16.27	0.63
NASA	37.55	35.04	2.51	9.07	4.38	4.69
PUKA	30.98	27.53	3.45	17.21	19.4	-2.19
Q142	22.84	20.65	2.19	5.32	5.01	0.31
Q027	35.35	34.42	0.93	7.51	13.14	-5.63
Q080	25.35	24.41	0.94	7.19	5.94	1.25
WIKD	35.98	34.73	1.25	12.83	9.7	3.13
ZDCO	28.16	27.22	0.94	11.89	11.26	0.63
AVERAGE	30.66	28.91	1.75	10.83	10.48	0.35

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VITA

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