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The Effect of Harvesting on Mangrove Forest Structure and the Use of Matrix Modelling to Determine Sustainable Harvesting Practices in South Africa

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1. Introduction

Mangrove forests exist along a transitional boundary between land and sea. They represent a continuum of biotic communities between terrestrial and marine environments (Hogarth, 1999; Kathiresan and Bingham, 2001; Alongi, 2008). These forests are globally distributed between the subtropical and tropical latitudes, restricted by major ocean currents and the 20°C isotherm of seawater in winter (Hogarth, 1999; Alongi, 2009). On a global scale, temperature is an important limiting factor but on regional and local scales variations in rainfall, tides, waves and river flow have a substantial effect on distribution and biomass of mangrove forests (Alongi, 2009). Erosion and depositional rates are also important as these affect the physical habitat that mangroves occupy. Generally the habitat of mangroves begins at mean sea level and extends to the spring high tide mark i.e. they exist in tidal areas (Hogarth, 1999; Spalding et al., 2010) while in South Africa mangroves are confined to estuaries that either may be permanently open to the sea or have an intermitted connection to the sea (Rajkaran, 2011). Estuaries are defined as “a partially enclosed body of water which is either permanently or periodically open to the sea and within which there is a measurable variation of salinity due to the mixture of seawater with freshwater derived from land drainage” (Day, 1980) as being; river mouths, estuarine bays, permanently open estuaries, temporarily open closed and estuarine lakes. There are five types of estuaries and these are defined by Whitfield (1992). The ecosystem services provided by mangroves include; shoreline protection from sea storms and excessive wave energy, nursery and areas of refugia for faunal populations (BOX 1), input of organic carbon into the food webs and filtration of silt and other compounds from the water column thereby protecting other nearshore ecosystems such as coastal reefs (Gilbert & Janssen, 1998; Fondo & Martens, 1998; Laegdsgaard & Johnson, 2001; Mumby et al., 2003). Mangrove forests are known to have survived for approximately 65 million years and therefore are resilient to large scale disturbances (Alongi, 2009). Key mangrove features that have assisted in their resilient nature include; the presence of a large reservoir of below-ground nutrients so that if a disturbance takes place the remaining nutrients will assist with the re-establishment of new seedlings to replace those that have been lost encouraging re-population of the disturbed area. Rapid biotic turnover has been recorded in mangrove forests and is facilitated by rapid rates of nutrient flux and microbial decomposition. Internal recovery after a disturbance is

accelerated by complex and efficient biotic controls such as nutrient-use efficiency (Alongi, 2008, 2009). Frequent, small scale disturbances such as harvesting disrupts the flow of nutrients from the living biomass to the sediment environment via the roots, it also facilitates changes to the microenvironment which will reduce the capacity of the mangrove forests to recover.

BOX 1.

Faunal diversity in mangrove forests is high including organisms from sponges to elasmobranchs and bony fish as well as bird species such as the Mangrove Kingfisher (Nagelkerken et al., 2008). Crabs are the most abundant macrofauna (numbers and biomass) in mangrove forests (Smith et al., 1991). They consume or hide 30 to 80 % of leaves, propagules and other litter on the floor of mangrove forests (Dahdouh-Guebas et al., 1997; Machiwa & Hallberg, 2002; Skov et al., 2002). Crabs enhance degradation of leaves and make the leaves available to meiofauna (Dahdouh-Guebas et al., 1999). The diversity of crabs found in a mangrove forest may vary. At Mngazana Estuary the following species were found *Neosarmatium meinerti* de Man, *Sesarma eulimene* de Man, *Sesarma catenata* Ortmann, *Uca lacteal annulipes* H. Milne Edwards, *Uca chlorophthalmus chlorophthalmus* (H. Milne Edwards), as well as *Parasesarma leptosome* (Hilgendorf) (Plate 1). The latter is a tree climbing crab that spends most of its life in the mangrove trees and is therefore totally dependent on mangrove forests for their existence (Emmerson et al., 2003; Emmerson & Ndenze, 2007). More recently the species *Perisesarma samawati* Gillikin & Schubart, which was only described to occur in East Africa was spotted at Mngazana Estuary in South Africa in 2011 for the first time (Plate 2).



Plate 1 and 2: Images of crab species only associated with mangrove forests. Photos taken by Anusha Rajkaran

2. Mangrove forests: Utilization and destruction

In 2003, the global estimate of mangrove forest cover was 14 650 000 ha and accounted for approximately 0.7% of the total global area of tropical forests (Wilkie & Fortuna, 2003; Giri et al., 2011). Each hectare is valued at between 200 000 – 900 000 USD (Wilkie & Fortuna, 2003; Giri et al., 2011). Human disturbances has resulted in more than 50% of the world's mangrove forests being destroyed (Spalding et al., 2010). This huge loss of mangrove forests globally, has been attributed to urban development, aquaculture, mining along coastal zones and overexploitation of fauna and flora of mangrove forests (Walters, 2005; Walter et al., 2008; Kairo et al., 2008; Alongi, 2009). The connection between coastal developments, water level fluctuations and mangrove loss or transformation has been recorded by a number of authors in South Africa and other parts of the world (Moll et al., 1971; Begg, 1984; Bruton, 1980;

Dahdouh-Guebas et al., 2005) (BOX 2). Worldwide, mangrove forests are harvested for a variety of purposes. The products are particularly important to subsistence economies, providing firewood, building supplies and other wood products (Bandaranyake, 1998; Ewel et al., 1998; Cole et al., 1999; Kairo et al., 2002; Dahdouh-Guebas et al., 2004, Walters et al., 2008). The subsequent effects on the ecosystem ranges from loss of habitat for fauna such as arboreal crabs (Emmerson and Ndenze, 2007), decreases in organic carbon export to the food webs and nearshore environments (Rajkaran & Adams, 2007), coastal erosion (Thampanya et al., 2006) and in the long term, loss of nursery functions (Laegdsgaard & Johnson, 2001).

BOX 2

Freshwater abstraction and poor bridge design has caused the mouths of some South African estuaries with mangroves to close to the sea more frequently, leading to long term inundation of roots and subsequent death of the mangroves (Breen & Hill, 1969; Bruton, 1980; Begg, 1984). Rising water levels have been one of the main factors that have lead to localised mangrove disturbances and mortalities in Kosi Bay (1965-1966) and Mgobezeleni Estuary (74 km south of Kosi Bay) (Bruton, 1980). Past data shows that 78% of the 1084 trees died in the Mgobezeleni Estuary due to submergence of the root structures when the water level rose for an extended period of time. This was a result of water being impounded behind a bridge constructed in 1971. Dead mangrove trees ranged from 40 cm to 15 m in height showing that all height classes are susceptible to death due to water level increases. The living mangrove stand became infested by the mangrove fern. In 2007, 77 *Brugueira gymnorrhiza* trees were still living, these have all since died (2011). The water level was ~ 30 cm of water above the sediment. Less than five seedlings were seen in areas where the sediment was not submerged. This estuary is a prime example of how poor coastal planning and developments can have a negative effect on surrounding coastal habitats such as mangrove forests.



Plate 3 and 4: Images taken at Mgobezeleni Estuary in 2007 by Dr. Ricky Taylor showing the submergence of the root structures of the *Bruguiera* trees and the extent of the mangrove fern.

2.1 Effect of harvesting on mangrove forests

Gaps created during the harvesting of either individual or groups of trees provide opportunities for seedling recruitment and growth (Rabinowitz 1978; Ewel et al., 1998; Sherman et al., 2000). The size class structure of mangrove forests in localities that experience harvesting show under-representation in large size classes, which is the result of selective harvesting (Saifullah et al., 1994; Walters 2005). Because mangrove wood is used for building, the size of the mangrove poles determines the role they play in the built structure. A comparison of height classes of the non-harvested and harvested sites in the Mngazana Estuary (31°42'S, 29°25' E) in South Africa showed that the height class 2.3 – 3.3 m was dominant in non-harvested sites while in harvested sites smaller trees were dominant. All the harvested poles were approximately 3 m (Rajkaran & Adams, 2010). Traynor & Hill (2008) interviewed harvesters with regard to harvesting preferences at Mngazana Estuary; they stated that any tree greater than 2 m in height with a desired diameter at breast height (DBH) would be harvested. They also stated that the required length of the wall poles used for building homesteads was 3 m for wall poles while roof poles were usually 4 m. This explained the differences found for mangrove height between harvested and non-harvested sites. Traynor & Hill (2008) recorded that the preferred species for building was *Rhizophora mucronata* (41% of participants preferred this species) and *Bruguiera gymnorrhiza* (21%) while *Avicennia marina* was used for firewood.

3. The use of matrix modelling to determine sustainable harvesting practices

With the use of population models one can predict the quantitative changes in population structure and thus add value to any management plan established for a particular mangrove forest. Mathematical models are popular conservation and management tools used to predict changes to plant and animal populations that are at risk due to activities such as harvesting (Raimondo & Donaldson, 2003; López-Hoffman et al., 2006; Owen-Smith, 2007; Ajonina, 2008). Matrix models are age or stage structured models used in cases when harvesting of particular size classes is the main risk. One takes into account the probability of an individual plant moving from one size class to the next i.e. transition probabilities as well as the possibility of the individuals persisting in the size class or dying (Caswell, 2001; Porte & Bartelink, 2002; Boyce et al., 2006; Owen-Smith, 2007; Caswell, 2009). In the case of plants, the model usually uses plant size (height or DBH) as the basis for the model. Model parameters include recruitment (the portion of propagules that is produced by a specific size class that is added to Size Class 1), mortality (M), transition rates (T) and persistence rates (P) for each size class, these are known as the vital rates (Caswell, 2001; Porte & Bartelink, 2002; Owen-Smith, 2007) (Figure 1).

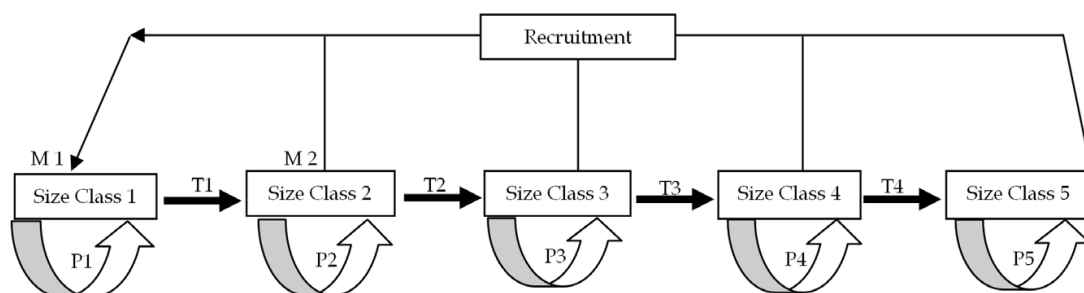


Fig. 1. The layout of the matrix model illustrating the vital rates mortality (M), transition rates (T) and persistence rates (P) for each size class.

The objective of this study was to develop a matrix model to determine the effect of different harvesting intensity scenarios, on the population structure of three mangrove species: *Avicennia marina*, *Bruguiera gymnorrhiza* and *Rhizophora mucronata*. The model results were compared to the observed population structure measured in the field at the end of the study in 2009 to determine the accuracy of the model and used to determine the most sensitive size classes to changes in vital rates within the population. Some data are presented here but more detailed results can be found in Rajkaran (2011).

3.1 Model development and accuracy

Nine sites at Mngazana Estuary were studied to collect data for the population model. This estuary is located in the Eastern Cape Province of South Africa, (Figure 2). In each site the following information was recorded, number of saplings (no hypocotyl less than 1 m), number of adults (over 1 m), the height of saplings and DBH and height of adults were measured. Subsequent measurements took place in November 2005, June 2006, November 2006, June 2007, November 2007, November 2008 and November 2009.

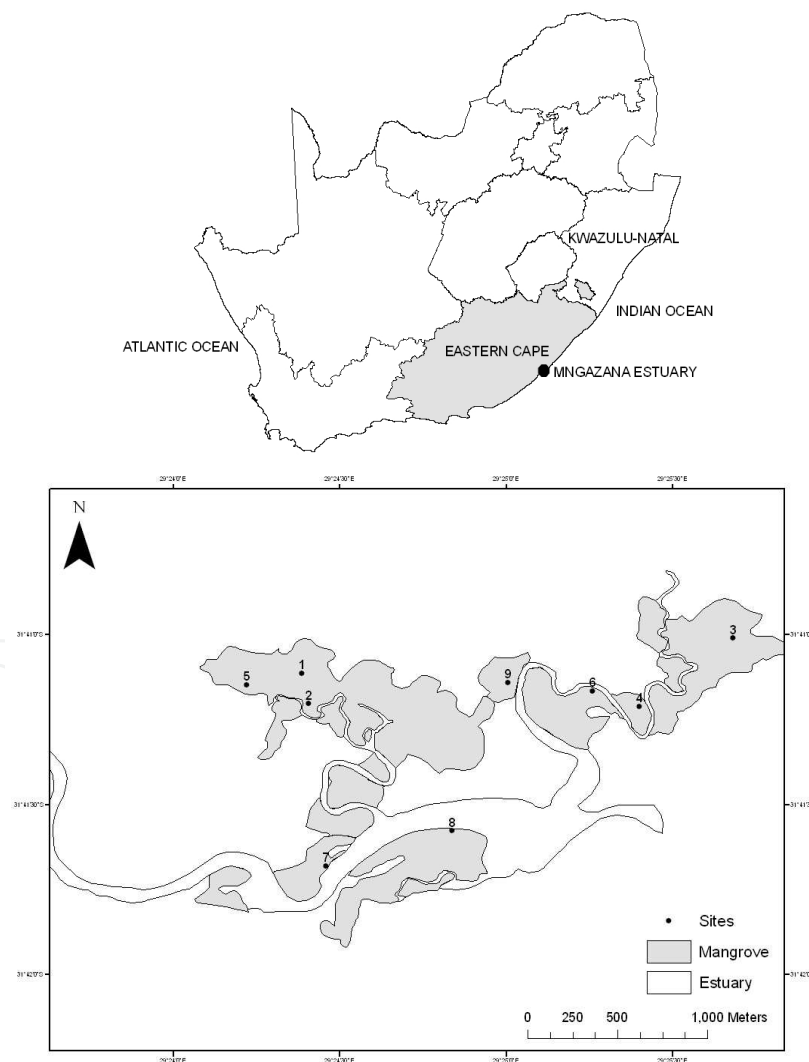


Fig. 2. The location of Mngazana Estuary in the Eastern Cape of the Republic of South Africa and the location of Sites 1-9 where growth was monitored from 2005-2009.

The population of each species, as calculated from nine sites around the estuary, was summarised and divided into a number of size classes based on mangrove height (Table 1). **Transition rates** were determined by counting the number of individuals in each size class over a period of five years (2005-2009). The **persistence rate** was the percentage of individuals that were in the same size class between two successive years (2005 compared to 2006). The transition rate was the percentage of individuals that were still alive but were now in the next successive size class therefore they had grown taller. Mortality rates were determined for the first two size class i.e. <50 cm and 50.5-150 cm height. The natural mortality of the other size classes could not be determined as none of the taller trees died unless they were harvested by the local community. In the model, natural mortality was included within the persistence rate i.e. the persistence rate was lowered by the appropriate percentage determined for each species based on the five year dataset. On two sampling trips (November 2005 and June 2006) the number of propagules on each tree was counted and the height of the tree was recorded. These data were used to determine the **fecundity** of each size class and were used as input on the proportion of propagules added by each size class to the total number of propagules.

Natural recruitment which was the number of new seedlings (hypocotyls present - <50 cm) added to the population was calculated for the five year period. Not all propagules that are produced establish themselves due to crab predation and removal by tidal movement. The number of individuals in each size class was converted from trees. m⁻² (calculated from site data) to trees.ha⁻¹. The number of individuals that an area is able to support (carrying capacity) was assumed to be the total number of individuals in the population. The model was formulated to be density dependant, therefore the greater the number of individuals in the total population the stronger the effect of competition on the smaller individuals resulting in a lower survival rate. The time span for each population model was determined by how long the population size would take to stabilise. N_t is the size of the population at the start of the study. N_{t+1} is the sum of all the size classes calculated for each year after the start of the study (t+1). The ratio between N_{t+1} / N_t is the finite rate of increase and summarises the dynamics of a population. This ratio is symbolised by lambda (λ -the dominant eigenvalue of the matrix). When $\lambda=1$ then the population is in balance and remains stable ($N_{t+1} = N_t$), if $\lambda>1$ the population is increasing ($N_{t+1} > N_t$) and if $\lambda<1$ then the population is decreasing ($N_{t+1} < N_t$) (Slivertown & Charlesworth, 2001; Rockwood, 2006). Initial model results were compared to the observed population structure measured in the field at the end of the study in 2009 to determine the accuracy of the model.

3.2 Harvesting intensity scenarios

Harvesting scenarios represented a static harvesting rate of 1, 5, 10, 15, 20 and 100% of individuals for the three different species present at Mngazana Estuary. To determine the effect of harvesting on the total population (N) as well as different size classes a number of harvesting scenarios were added to the model. Population monitoring showed that harvesting of trees taller than 250 cm was common, therefore the model assumed that a percentage of Size Class 4 (250-350 cm) and 5 (>351 cm) would be harvested each year. The following harvesting intensities were used; 1, 5, 10, 15, 20 and 100% of a particular size class.ha⁻¹.year⁻¹. These scenarios would show how much of the population could be harvested and what the limit was for harvesting. The scenarios also showed how each size class changed in abundance in response to the different harvesting intensities.

3.3 Results

The *Avicennia marina* trees at Mngazana Estuary are either completely harvested or portions of the tree are cut for firewood. The assumptions for this model were 1) a tree, or portion of a tree, used for firewood is taken as a completely harvested tree and 2) that harvesting only affects the tallest trees in the forest (S5). The second assumption was based on field observations from Mngazana and Mhlathuze estuaries, where the tallest trees were the ones that were targeted. A hundred percent harvesting of individuals in the tallest size class decreased the total population to below 10 000 trees.ha⁻¹ (Figure 3) and λ to 0.994 (Table 2). Restricting harvesting to just one size class that has reached reproductive maturity will ensure that other trees will still be present to produce propagules and subsequently seedlings. For this reason λ values as shown in Table 2 for *Avicennia* remain just below 1 for all harvesting scenarios. The number of individuals in Size Class 2 under 0% harvesting stabilised at less than 10 000 per ha (Figure 4). This decreased when the harvesting intensity increased as did the number of individuals in all size classes. To ensure more than 5 000 individuals were present in Size Class 1, which represents the main class for natural regeneration, harvesting must not exceed 20% of the trees taller than 350 cm per year. This is equivalent to 238 ± 4.5 harvested trees.ha⁻¹.yr⁻¹.

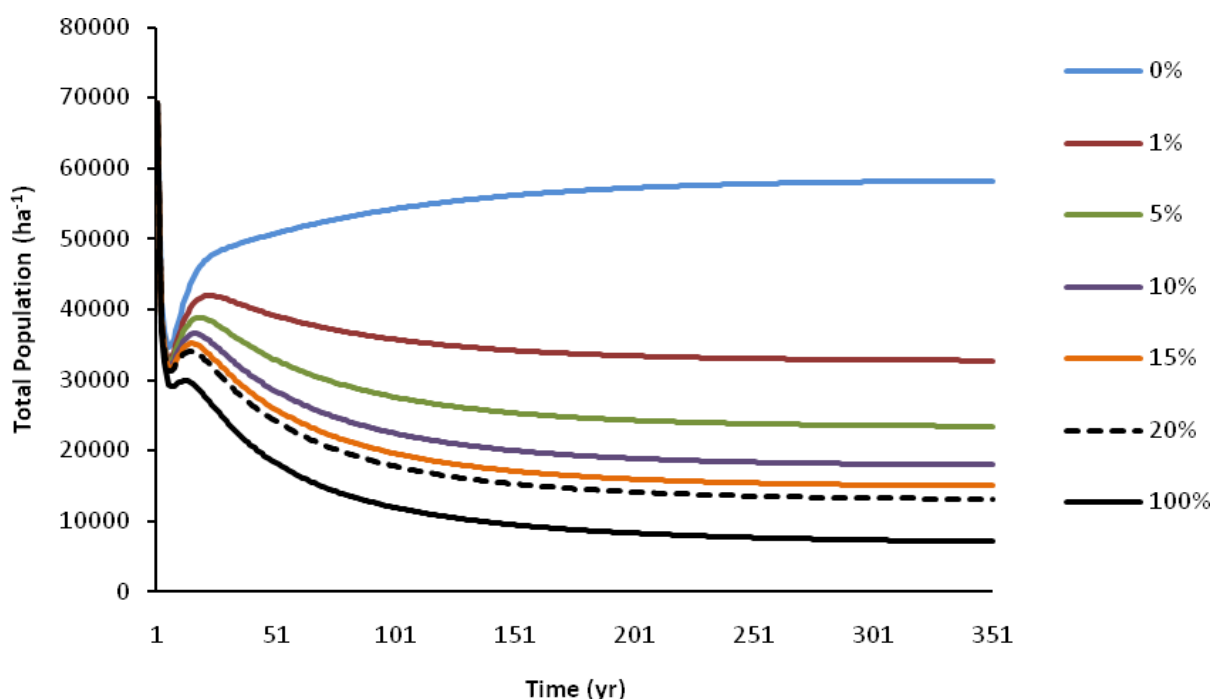


Fig. 3. Changes in total population size for the species *Avicennia marina* over time in response to different harvesting scenarios.

The assumption was that harvesting of two size classes would take place at Mngazana Estuary for *Bruguiera*. All trees greater than 251 cm would be removed. Harvesting of this species had a dramatic effect on the total population size. The total population of this species decreased by 63% when harvesting intensity was set at 1%. This allowed the population to stabilise at 15 000 trees.ha⁻¹ (Figure 5). A further scenario was run using a harvesting intensity of 2%, this reduced the total population to approximately 5 000 trees.ha⁻¹. The mean λ for this species dropped from 0.999 to 0.834 at 100% harvesting intensity showing that the population was decreasing and natural regeneration was not taking place (Table 3).

Species	Size class (Height)	S1 <50 cm	S2 50-150 cm	S3 151-250 cm	S4 251-350 cm	S5 >351 cm
<i>Avicennia marina</i>	$N_{(t0)}$ (per ha ⁻¹)	16 786	40 536	8 036	2 500	1 339
	T	0.2	0.1	0.1	0.1	0
	P	0.6	0.8	0.9	0.9	0.9
	F	0	0	0	0.5	0.5
	MR (%)	21.0 ± 6.8	6.9 ± 2.0	ND	ND	ND
<i>Bruguiera gymnorhiza</i>	$N_{(t0)}$ (per ha ⁻¹)	12 831	10 703	2 109	2 188	2 266
	TR	0.08	0.08	0.12	0.02	0
	PR	0.79	0.8	0.88	0.98	0.9
	F	0	0.16	0.16	0.33	0.33
	MR (%)	12.2 ± 4.6	7.2 ± 7.6	ND	ND	ND
<i>Rhizophora mucronata</i>	$N_{(t0)}$ (per ha ⁻¹)	11 979	43 750	10 104	8 125	2 917
	TR	0.3	0.03	0.1	0.03	0.1
	PR	0.6	0.88	0.9	0.97	0.9
	F	0	0.16	0.16	0.33	0.33
	MR (%)	15.6 ± 3.6	8.5 ± 2.3	ND	ND	ND

Table 1. Summary of data for each species and size class (S1-S5) used to populate the matrix models. (Transition rates (T) and persistence rates (P), fecundity rate (F), mortality rate (MR)).

Harvesting intensity	Total Population (N)	Size class (Height (cm))				
		0-49	50-150	151-250	251-350	>350
0%	1.000	1.001	0.997	1.001	1.004	1.006
1%	0.998	0.999	0.996	0.999	1.002	1.004
5%	0.997	0.998	0.995	0.998	1.002	1.002
10%	0.996	0.997	0.994	0.998	1.001	1.001
15%	0.996	0.997	0.994	0.997	1.001	0.999
20%	0.996	0.996	0.994	0.997	1.000	0.998
100%	0.994	0.995	0.992	0.996	0.999	0.999

Table 2. Mean λ values for *Avicennia marina* under different harvesting scenarios after 350 years.

Harvesting intensities of 15% and 100% were omitted from the graphs as the curves were similar to the 20% harvesting intensity and were not visible. Harvesting 1% of the adult trees maintained the density of size class 1 to < 5 000 individuals.ha⁻¹ (Figure 6).

The same assumption regarding harvesting was used for *Rhizophora mucronata* that harvesting of two size classes would take place at Mngazana Estuary. Documented data showed that the average length for harvested poles was 3.4 m. Harvesting scenarios in the model were restricted to the last two size classes (>251 cm). Total population size decreased from ~ 80 000 to 28 000 individuals.ha⁻¹ when harvesting intensity was 1%, this represented a 65 % reduction (Figure 7). λ values decreased to less than 1.000 showing that the population was decreasing as a result of the harvesting (Table 4). Harvesting intensity greater than 15% decreased the density of Size class 1 to ~3 500 individuals.ha⁻¹ (Figure 8). Harvesting between 5-10% of trees per year would amount to 183 – 283 harvested trees.ha⁻¹.yr⁻¹.

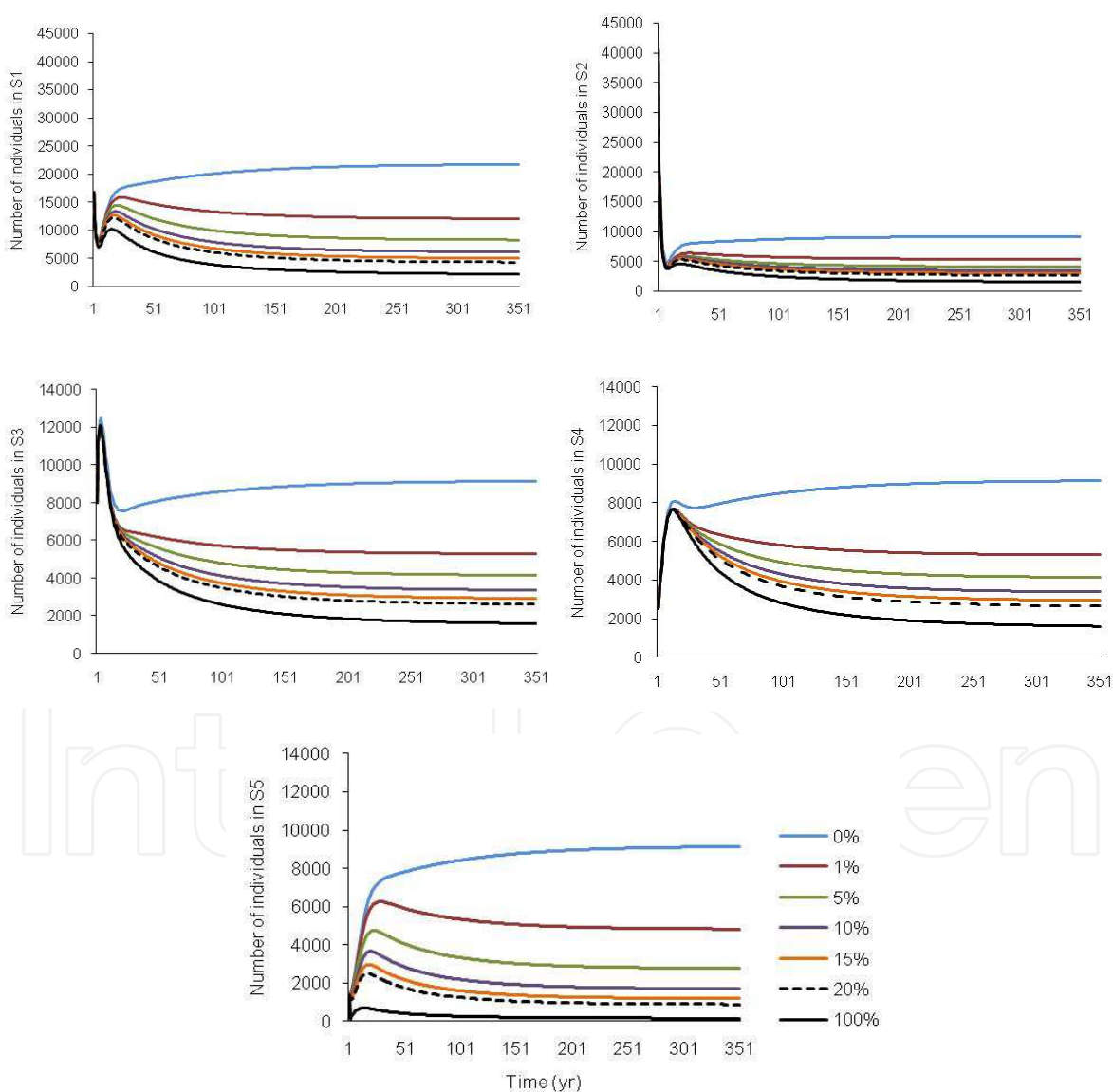


Fig. 4. The impact of harvesting on the number of individuals.ha⁻¹ in each size class of the *Avicennia marina* population over time. (Y-axis was not standardised for all graphs so that curves would be visible)

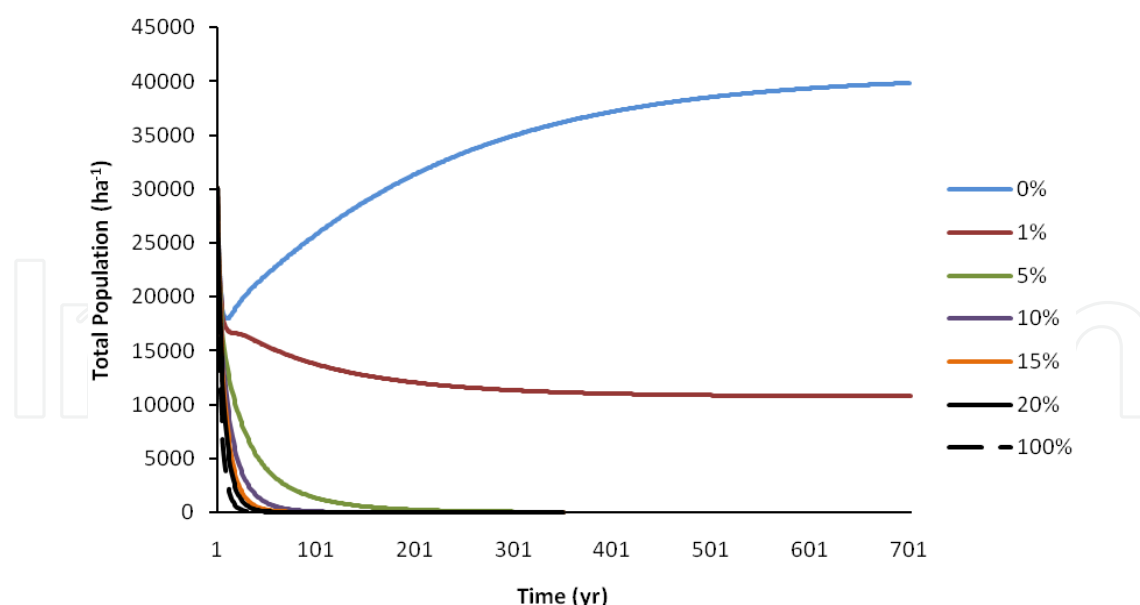


Fig. 5. Changes in total population number for the species *Bruguiera gymnorrhiza* over time in response to different harvesting scenarios.

Harvesting intensity	Total Population (N)	Size class (Height (cm))				
		0-49	50-150	151-250	251-350	>350
0%	1.000	1.001	0.999	1.000	1.002	1.000
1%	0.999	0.999	1.000	1.000	0.999	0.999
5%	0.980	0.980	0.979	0.981	0.983	0.977
10%	0.949	0.949	0.948	0.950	0.951	0.946
15%	0.922	0.922	0.922	0.924	0.924	0.918
20%	0.901	0.901	0.901	0.902	0.902	0.896
100%	0.834	0.832	0.835	0.834	0.832	0.822

Table 3. Mean λ values for *Bruguiera gymnorrhiza* for different harvesting scenarios after 701 years, the number of years required for the population to reach equilibrium was greater than for the other two species.

3.4 Discussion

Small scale disturbances such as harvesting, depending on the timing, frequency and intensity, which result in the loss of some of the mangrove population, may lead to natural regeneration if there are existing seedlings, saplings and mother trees (standard) around the disturbed area, if there is potential for water-borne propagules to travel to the area via tidal flow and if the propagules from disturbed trees are still present (FAO, 1994). A “standard” is defined as a seed bearing tree that can withstand exposure to strong winds and light and, in fringe areas, high tidal action (FAO, 1994). Regeneration will be restricted if the number of standards is reduced, if dead trees and branches reduce the light on the forest floor, if damage occurs to surrounding seedlings/saplings due to trampling and if a substantial change in soil conditions occurs (FAO, 1994; Harun-or-Rahsid et al., 2009).

Clarke et al., (2001) noted that the lack of diaspore dormancy in most mangrove species translates into a small or non-existent seed bank. The lack of a persistent soil seed bank of

true mangrove species decreases the probability of a full recovery by mangrove populations after large scale disturbances and increases the chances of invasions of mangrove-associate species (Dahdouh-Guebas et al., 2005; Harun-or-Rahsid et al., 2009). Populations are reliant on regular cohorts of diaspores for regeneration so their continuous production by adults is vital. Rajkaran & Adams (2007) recorded movement of propagules out of the creeks and main channel of Mngazana Estuary, dispersed propagules were found on the adjacent beach near the mouth of the estuary. At Mngazana Estuary the presence of propagules on the forest floor is dependent on that produced by the adults in that specific area and not on the propagules brought in by tides. So at this estuary the continuous production by adults remaining after harvesting is vital for natural regeneration.

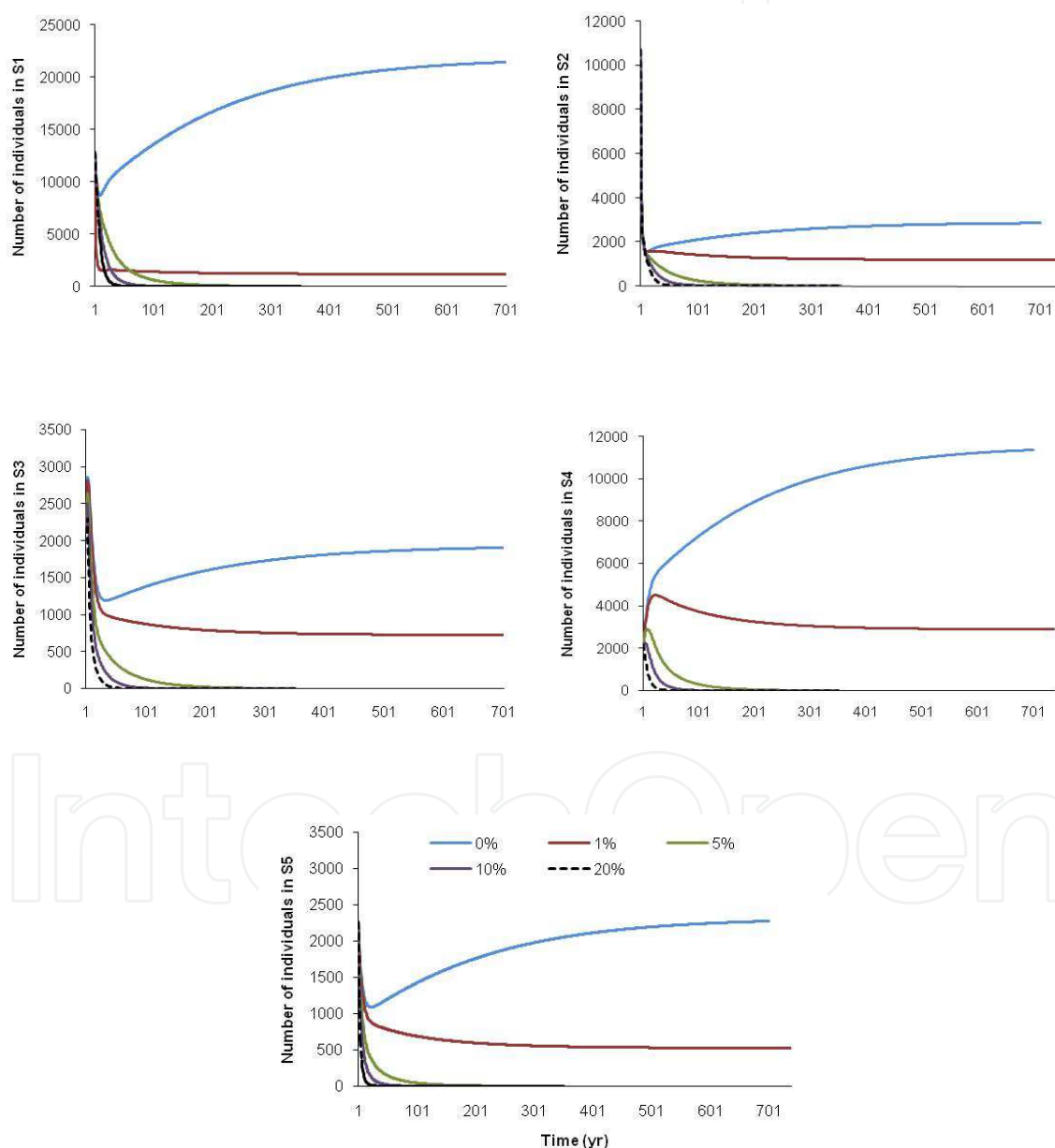


Fig. 6. The impact of harvesting on the number of individuals.ha⁻¹ in each size class of the *Bruguiera gymnorhiza* population over time. (Y-axis was not standardised for all graphs so that curves would be visible).

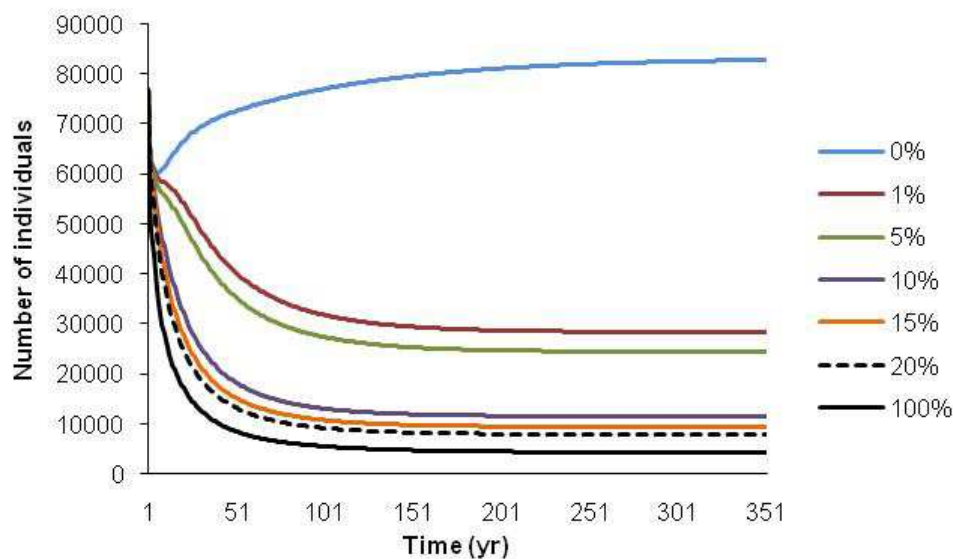


Fig. 7. Changes in total population size for the species *Rhizophora mucronata* over time in response to different harvesting scenarios.

Harvesting intensity	Total Population (N)	Size class (Height (cm))				
		0-49	50-150	151-250	251-350	>350
0%	1.000	1.003	0.998	0.999	1.003	1.002
1%	0.997	1.000	0.996	0.997	0.997	1.000
5%	0.997	0.999	0.996	0.996	0.997	0.999
10%	0.995	0.997	0.994	0.995	0.993	0.994
15%	0.994	0.996	0.994	0.994	0.992	0.993
20%	0.994	0.996	0.994	0.994	0.992	0.991
100%	0.992	0.994	0.992	0.992	1.001	0.974

Table 4. Mean λ values for *Rhizophora mucronata* for different harvesting scenarios after 350 years.

Size classes in this study were based on height as previous studies have shown that harvesters targeted specific heights within the population (Rajkaran & Adams, 2009; Traynor & Hill, 2008). A density dependent model was used to simulate population structure and growth over time and the results conformed well to the logistical equation. The average λ value for each species in the absence of harvesting scenarios was 1.000, which shows that the populations are not increasing under the current harvesting rates for each size class. This may be a consequence of the continuous past harvesting in the Mngazana mangrove forest that has influenced vital rates. This was not taken into account in this model. López -Hoffman et al. (2006) recorded λ values of 1.050 when no harvesting was taking place. Vital rates for *Rhizophora mucronata* were comparable to those measured by López -Hoffman et al., (2006). Persistent rates ranged from 0.909 to 0.983, while transition rates ranged from 0.026-0.034 for adult size classes in that study, which is similar to the current study for this species. Similar studies for *Bruguiera gymnorrhiza* were not found. Clarke, (1995) used a matrix model to predict the population dynamics of *Avicennia marina* in New Zealand. Persistence rates for seedlings were 0.825, saplings - 0.909, young tree -

0.963 and older trees 0.999, while transition rates were 0.010, 0.073, 0.008, 0.012, 0.000 respectively. Sizes of each life stage were not stated in the study. The persistence rate in this study for *Avicennia* seedlings was much lower at 0.6 and transition was higher at 0.2, while all other rates were comparable with other studies. This implies that *A. marina* seedlings in South Africa grow faster and more seedlings survive to the next population size class within one year but the overall survival of the seedlings is similar between the two studies. Faster growth rates are dependant on site specific environmental conditions such as sediment characteristics and interspecific competition (Rajkaran, 2011).

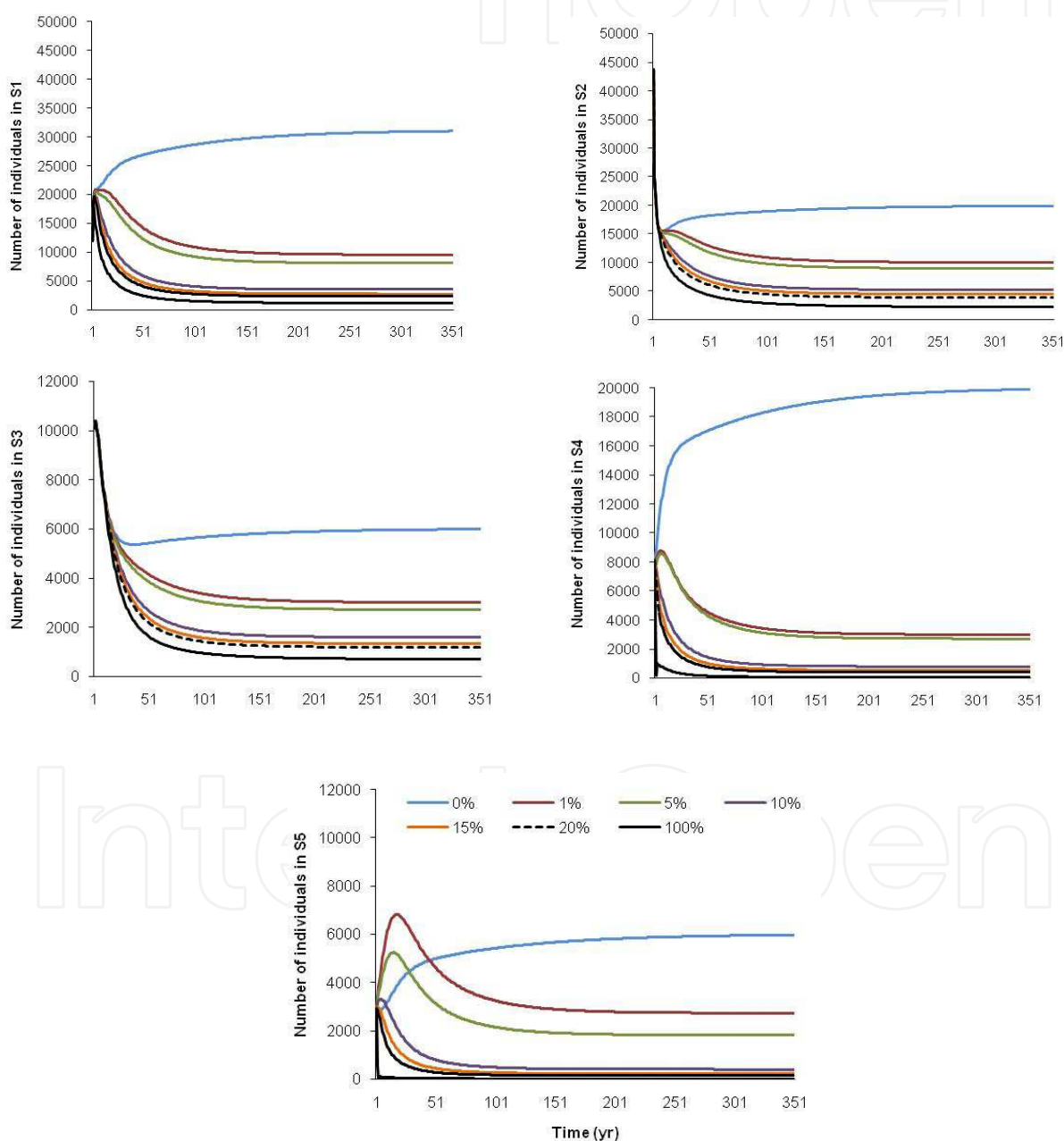


Fig. 8. The impact of harvesting on the number of individuals.ha⁻¹ in each size class of the *Rhizophora mucronata* population over time. (Y-axis was not standardised for all graphs so that curves would be visible for S5 and S3)

All harvesting scenarios decreased λ to less than 1.000, showing that the populations were decreasing in size. A sustainable harvesting rate would be one where λ is greater than 1. This would indicate that harvesting would be increasing the population growth by increasing space and decreasing competition between individuals. A λ value of 1.000 would mean that the population is unchanging (López -Hoffman et al., 2006) and disturbance would be detrimental to the population. FAO (1994) have set minimum limits for the number of “mangrove” seedlings that must be present to facilitate natural regeneration once adults have been removed from the population. The harvesting intensity that leads to a seedling density of less than 5000.ha⁻¹ were 100% intensity for *Avicennia marina* all intensities greater than 1% for *Bruguiera gymnorrhiza* and 15, 20 and 100% for *Rhizophora mucronata*. The limits of harvesting in the Mngazana mangrove forest should not approach these levels. López -Hoffman et al., (2006) set sustainable harvesting in the Río Limón mangrove forests of Lake Maracaibo in Venezuela at 7.7% per year for *Rhizophora mangle*, the current study has set harvesting limits at 5% per year for *Rhizophora mucronata* and *Avicennia marina*. Harvesting of *Bruguiera gymnorrhiza* should be stopped as the density of this species is lower than the other two species. Preferably there should be no harvesting of this species. Harvesting intensity must ensure that seedling density is maintained within acceptable limits as set out in the published literature (FAO, 1994; Bosire et al. 2008; Ashton & Macintosh, 2002). A density of 2 500 - 3 200 seedlings ha⁻¹ has been suggested as a minimum number required for natural regeneration to take place after a disturbance (FAO, 1994; Bosire et al. 2008). Ashton and Macintosh (2002) recommended 5 000-10 000 seedlings ha⁻¹ for adequate regeneration in a cleared area in the Matang Mangrove forest in Peninsular Malaysia. Density of individuals of the three species were measured at Mngazana Estuary in 2005 and were found to be 17 000, 13 000 and 12 000 seedlings.ha⁻¹ for *Avicennia*, *Bruguiera* and *Rhizophora* respectively. To set the minimum number of seedlings to 5 000 individuals.ha⁻¹ would mean that this size class would be more than half the original density. Increasing the limit to 10 000 seedlings.ha⁻¹ would be more acceptable at the Mngazana Estuary for all species. The harvesting limits for each species will be different but managers must ensure that the seedling densities are maintained.

Mangrove management regimes may also suggest different densities for standards, i.e. the reproductively active trees producing propagules; these range from 7 (Malaysia) to 20.ha⁻¹ (Phillipines) (Choudhury, 1997). This depends on the species; FAO (1994) suggested 12 standards.ha⁻¹ for the genus *Rhizophora*. These levels are recommended for forests where clear-felling takes place in tropical countries where growth rates are high. Clear felling should be avoided in the Mngazana mangrove forest as this will significantly change sediment characteristics. Sediment conditions are significantly affected by changes in vegetation cover and plant density in a mangrove forest (Rajkaran and Adams, 2010). Mangrove forests are made up of species that are able to attain slow growth under a wide variety of conditions (Krauss et al., 2008) but Rajkaran and Adams (under review) recorded that growth and mortality of different size classes within a population were related to certain sediment parameters i.e. seedling growth was negatively related to high sediment pH (*Rhizophora* upper limit for pH in this study was 7.1) while seedling mortality for *Bruguiera* was negatively affected by an increase in sediment moisture.

A harvesting intensity of 5 % would maintain the number of individuals for *Rhizophora mucronata* at greater than 3 000.ha⁻¹ in Size Class 3 and Size Class 4 while Size Class 5 would be reduced to approximately 2 000 individuals.ha⁻¹. Traynor & Hill (2008) estimated the annual demand for mangroves at 18 400 stems.yr⁻¹ at Mngazana. These

were mainly used by the local communities to build homesteads. The suggested harvesting intensity of between 5 and 10% per year would provide this required number of stems and indeed yield more harvested stems than those required at the time of the 2008 study. A more detailed study about the increase in the demand over time due to increases in the human population is required, but in the meanwhile an alternative wood resource must also be provided to the communities to replace the mangroves. The full effects of harvesting have not been measured in this study because, for example, the effects of trampling on seedling survival and its influence on population growth and structure were not addressed. Recruitment was extremely low in this study which may have been the influence of physical disturbance from harvesters. Other management recommendations include reducing harvesting within the 10 - 20 m strip from the estuary channel. The purpose would be to sustain trees that form a barrier between the energy of the water flowing in on a high tide and the young seedlings.

4. Management of mangrove systems in South Africa

The management of ecosystems calls for the interaction between researchers and society to ensure that environmental and socio-economic issues are integrated with government policies. For this to take place a number of conceptual frameworks exist as tools for communication between researchers and end users of environmental information such as government departments (Maxim et al., 2009). The Drivers-Pressures-Status-Impact-Response (DPSIR) framework focuses on the connecting relationships between the **Driving** forces that are usually societal and economic developments that place the environment under **Pressure** which alters the **State** of the environment, and **Impacts** on the ecosystems. The **Response** from society is usually in the form of regulatory laws or rehabilitation plans depending on the situation (Bidone & Lacerda, 2004; Maxim et al., 2009; Omann et al., 2009; Atkins et al., 2011). The DPSIR framework allows managers and scientists to highlight issues that must be prioritised with regard to management of natural systems. The DPSIR framework was applied to the results from this research and identifies the issues associated with the management of mangroves in South Africa (Figure 9).

Overall interventions for the conservation of mangroves in South Africa include directly protecting pristine mangroves, protecting the hydrological regimes supporting these ecosystems (particularly freshwater quantities flowing into the estuaries-which would be dependent on the base-flows required to maintain mouth conditions in the optimal state), promoting natural regeneration for self renewal, enforcing mangrove buffer zones and the continued capacity development and education of those communities that use the forests (Macintosh & Ashton, 2004). Mangrove buffer zones provide protection to any habitat or human areas behind them. Vietnam maintains a 100 m - 500 m wide belt of mangroves to protect the Mekong Delta coastline against storm and flood protection, while the Philippines maintain a 20 m wide zone for protection of shorelines (Macintosh & Ashton, 2004). All mangroves in South Africa are found within estuarine ecosystems so their capacity to protect the coastline is limited. However in many cases coastal developments have occurred along the banks of estuaries behind mangrove and salt marsh communities. In these cases it is recommended that a mangrove buffer zone of 25 m be maintained and in the case of creeks, a 10 m buffer zone should be created. No activities, such as harvesting, should take place within these zones. In addition to these measures the identification and promotion of alternative resources for building is required.

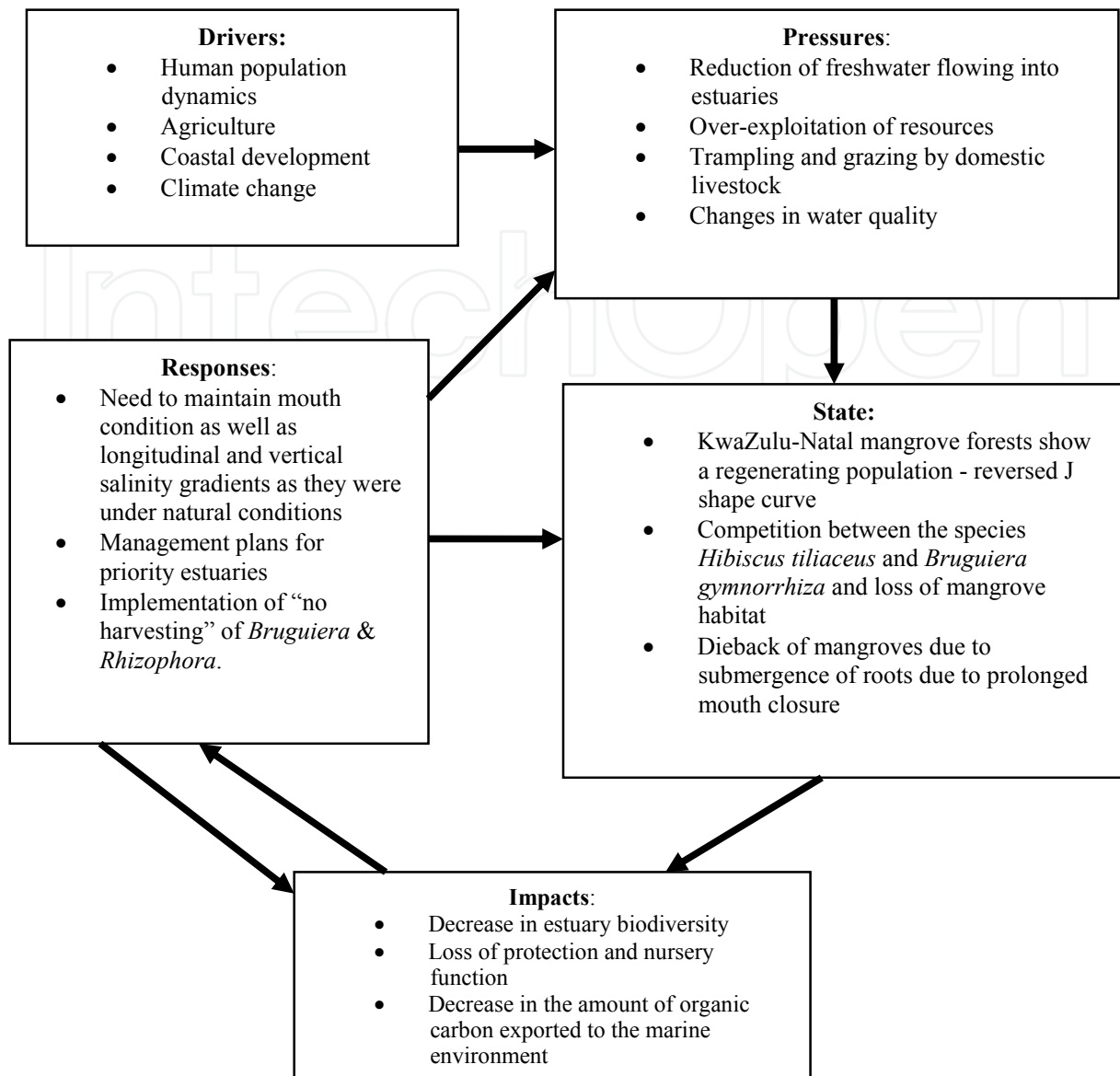


Fig. 9. Summary of DPSIR framework for the mangrove forests of South Africa.

5. Conclusion

Matrix modelling has allowed us to determine how much of a mangrove forest can be harvested while still maintaining a viable population. These data must be included in any management plan which includes the continual use of the forests as a wood resource for the local communities. The model presented here can be used by managers at other forests but growth data would need to be collected first as vital rates presented here will differ to other mangrove forests.

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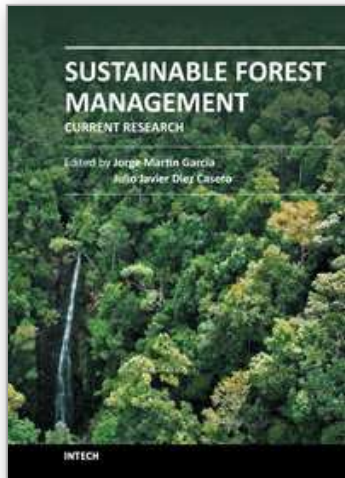
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Sustainable forest management (SFM) is not a new concept. However, its popularity has increased in the last few decades because of public concern about the dramatic decrease in forest resources. The implementation of SFM is generally achieved using criteria and indicators (C&I) and several countries have established their own sets of C&I. This book summarises some of the recent research carried out to test the current indicators, to search for new indicators and to develop new decision-making tools. The book collects original research studies on carbon and forest resources, forest health, biodiversity and productive, protective and socioeconomic functions. These studies should shed light on the current research carried out to provide forest managers with useful tools for choosing between different management strategies or improving indicators of SFM.

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