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Multi-temporal reconstruction of long-term changes in land cover in and around the Swartkops River Estuary, Eastern Cape, South Africa

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Abstract Multi-date remotely sensed images comprising Landsat TM images of 1984, 1993 and 2003 and, Landsat OLI images of 2013 were used to reconstruct long-term changes in land cover in the Swartkops River Estuary by mapping changes in vegetation distribution over a period of ~30 years between 1984 and 2013. These images were complemented by high-resolution near-anniversary aerial photographs that were used as ancillary sources of ground truth during supervised classification of the Landsat images. Results of our investigation point to human-induced loss of biodiversity due to persistent encroachment of different development activities on terrestrial vegetation, substantial expansion of the salt marsh due to climate change–driven relative sea level rise and persistent increase in keystone salt marsh vegetation species notably *Zostera capensis* and *Spartina maritima* due to the combined influence of human-induced nutrient loading into estuarine water and relative sea level rise. These observations argue for the immediate need to embrace appropriately informed management interventions in order to enhance the sustainability of salt marsh ecosystems for the benefit of present and future generations.

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Introduction

There are numerous definitions of what an estuary is, with different authors defining it within the context of specific settings, purpose and objectives of their research. Odu (1959) defines an estuary as 'a river mouth where tidal action brings about a mixing of salt and fresh water'. Day (1980) defines it as 'a partially enclosed coastal body of water which is either permanently or periodically open to the sea and situated at the interface between fresh river-water and marine water with measurable variation of salinity due to the mixture of sea water with freshwater derived from land drainage' while Pritchard (1967) defines it as 'a semi-enclosed body of water which has a free connection with the open sea and within which sea water is measurably diluted with fresh water derived from land drainage'. Although it is beyond the scope of this paper to provide an exhaustive overview of how different authors have tried to define what an estuary is, the simplest definition of an estuary is that it is an area where a river meets the sea to allow seawater and fresh river water to mix and create a marshy environment which is neither sea nor river.

Estuaries are important ecosystems because they provide (1) conduits for the transportation of sediments and nutrients into the marine zone, where they contribute to marine ecosystem productivity (Driver et al. 2004; Strydom et al. 2003), (2) nursery areas for marine fish by providing rich food supplies that favour rapid growth and protection from marine predators and vital habitats for several fish species that are depended on estuaries for their entire life cycles (Costanza et al. 1997; Wallace et al. 1984), (3) aisles through which species move between oceans and rivers and feeding and staging sites for significant populations of migratory birds (Whitfield 1998; Turpie 1995), (4) highly productive environments that contribute substantially to the production of inshore fisheries by supporting wide-ranging endemic species that depend on estuaries for their survival (Lamberth and Turpie 2003; Turpie et al. 2002), (5) aesthetically appealing environments in which to live and support different non-extractive recreational activities (Hosking 2011). They also provide wide-ranging plants that are often used for different purposes, business opportunities for people who provide support services and various products other than fish, i.e. worms that are used as bait by anglers and crabs and prawns that are harvested for food. Apart from providing these ecosystem goods and services, they also regulate the discharge of sediments, filter excess nutrients from terrestrial sources before river water mixes with sea water and dissipate the potentially destructive effects of high energy events such as floods and coastal storm surges. In South Africa, as is the case in other countries, estuaries contribute substantially to the country's economy with recent estimates placing the total value of estuarine fisheries at R433M compared with R490.4M for inshore fisheries (Lamberth and Turpie 2003).

Although estuaries are not in danger of disappearing or becoming extinct as happens to endangered species, they are vulnerable to natural and human-driven changes that often lead to loss of biodiversity and valuable ecosystem services (Driver et al. 2004) under increasing development pressure and the adverse effects of climate change (Van Niekerk and Turpie 2012). We decided to monitor long-term changes in land cover because they are capable of directing attention to qualities of the environment by providing visible expressions of how the environment is changing in ways that enable us to recognise the need for action by pointing to some of the major drivers of observed changes with the logic informing this reasoning being premised on the fact we need to know about patterns before we can understand processes (Underwood et al. 2000). This reasoning is articulated in the discussion by employing a framework of analysis that attempts to illustrate how (1) *Driving forces* in the form of climate change and human-environment interaction are exerting (2) *Pressures* on the environment that lead to wide-ranging changes in the (3) *State* of the environment and (4) *Impacts* that elicit or are potentially capable of prompting societal (5) *Responses* that create a vicious cycle of causally linked factors by feeding back into the driving forces in a manner that has come to be widely used to assess environmental challenges and policy responses in the form of the Drivers–Pressures–State–Impacts–Responses (DPSIR) framework (Gupta et al. 2019; Lewison et al. 2016; Oesterwind et al. 2016; Gregory et al. 2013; Bell 2012; Ness et al. 2009; Svarstad et al. 2007; Odermatt 2004; La Jeunesse et al. 2003; Walmsley 2002; Berger and Hodge 1998). Although we used the DPSIR framework in our analysis because of its established usefulness in interrogating complex environmental problems (Elliot et al. 2017; Hamandawana et al. 2005), we did this cautiously by acknowledging that usage of this framework is often confounded by inconsistent usage of the terms 'driver' and 'pressure' with some studies defining climate change as a driver for example (MA. 2005), while some authors, i.e. Omann et al. 2009 and Halpern et al. 2008 define it as a pressure and threat respectively.

Research methods

Study area

The Swartkops River Estuary (Fig. 1a) is located in the Nelson Mandela Bay Municipality in the Eastern Cape Province of South Africa.

It is a permanently open salt marsh estuary (Bornman et al. 2016) covering approximately 682 ha (Enviro-Fish Africa 2009) and is considered to be unique because of its positioning in a highly urbanised and industrialised area. It is surrounded by high and medium density residential areas that include the Swartkops Village, Redhouse and Amsterdamhoek/Bluewater Bay and situated in the vicinities of the townships of Kwazakele and Motherwell. Major industrial activities in the localities of this estuary include carbon black manufacturing from oil, motor vehicle manufacturing and the sale industrial accessories and spare parts, fish-water sewerage works, sand and clay mining, brick manufacturing, saltpans, tanneries, wool industries, railway yards and depots and subsistence bait fishing (Nel 2014; IMP:SE and SREVANR 2011). Although rainfall in this environment averages 636 mm/annum (Reddering and

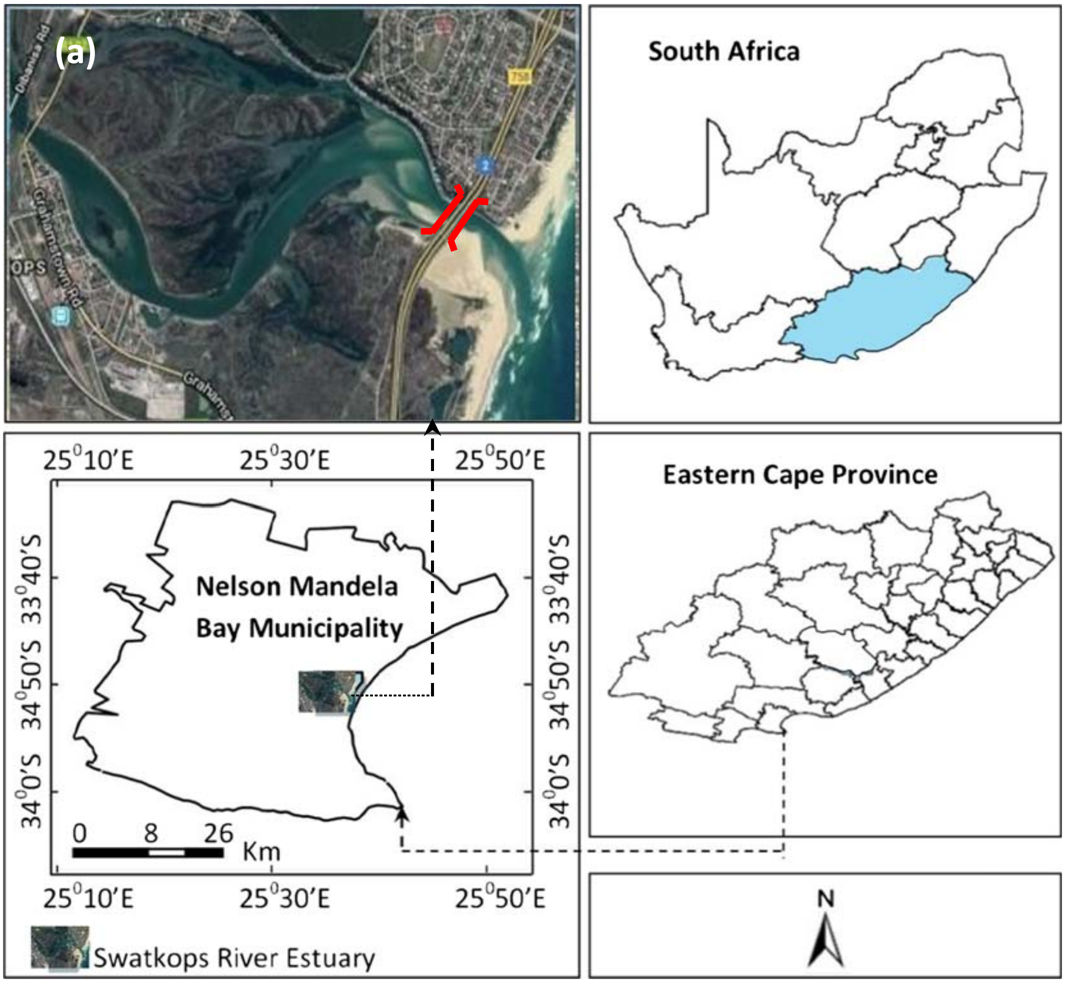

Fig. 1 Location of the Swartkops River Estuary (SRE)

Esterhuizen 1981), agriculture is very limited and largely confined to cattle, sheep and poultry and ostrich farming in the estuary's catchment areas (ZRWRMP 1999). The dominant types of estuarine vegetation include dense stands of *Phragmites australis*, *Zostera capensis* and *Spartina maritima* (Bornman et al. 2016; Nel 2014) with terrestrial vegetation along the estuary's fringes consisting of dense thickets which provide a vital habitat for the area's sedentary bird populations and reptiles that are becoming increasingly threated by the combined effects of climate change, encroachment of human settlement and a wide range of development activities.

Prominent natural habitat types include estuarine water, the floodplain saltmarsh and supratidal saltmarsh and sandbanks and mudbanks which are vital for resident and migrant bird species that include white-breasted cormorants (*Phalacrocorax lucidus*), sacred ibis (*Threskiornis aethiopicus*), Kelp Gull (*L. dominicanus*), grey-headed gull (*Larus cirrocephalus*), wimbrel, grey plover and curlew sandpiper (https://www.zwartkopsconservancy.org/conservation.html; Enviro-Fish Africa 2009). Man-made features over and within the estuary's immediate environs in addition to those listed above include roads and bridges, a dense informal settlement on the banks of the Chatty River, medium and low density residential areas with cattle and ship farming dominating the livestock sector (Bornman et al. 2016; Nel 2014). The estuary's main water supply source is the Swartkops River whose flow is augmented by seasonally variable inflow from numerous tributaries of which the Elands River and the Chatty River are the most active (Baird et al. 1986). Although it is the third largest saltmarsh estuary in the country (Enviro-Fish Africa 2009), it does not enjoy the status of a protected estuary (Turpie et al. 2002). The SRE is therefore a very important ecosystem for biodiversity conservation and deserves priority consideration because of the numerous ecosystem services and goods it provides (Colloty et al. 2001).

Image compilation

The datasets that were used in this investigation include dry season Landsat 5 TM images of 1984 and 1993, Landsat Enhanced Thematic Mapper (ETM) and Landsat Operational Land Imager (OLI) images of 2003 and 2013 respectively and like season high-resolution panchromatic and true colour aerial photo-mosaics of 1980, 1994, 2003 and 2013. Satellite images were downloaded from the US Geological Survey archives and the aerial photographs compiled from single frame coverages that were acquired from South Africa's Departments of Rural Development and Land Reform (DRDLR) in Cape Town and Pretoria. Table 1 describes the temporal sequencing and characteristics of these images.

Dry season Landsat images were purposefully selected in order to facilitate the acquisition of cloud-free coverages and identification of salt marsh vegetation under low flood conditions with like-season aerial photographs being preferred in order to enhance close temporal correspondence with Landsat images. The 2013 and 2003 aerial photographs were acquired georeferenced from source. These photographs were mosaicked in ERDAS IMAGINE to provide foot coverages of the study area that were used to georeference the remaining 1994 and 1980 aerial photographs to provide a complete set of aerial-photo mosaics covering all time slices. These mosaics were subsequently used to georeference their corresponding Landsat images by systematically pairing them so that each set consisted of pairs that were temporally closest to each other following the order of acquisition dates provided in Table 1. Thereafter, all mosaics were clipped to provide spatial coverage of a study area covering 460 ha out of the estimated 682 ha covered by the SRE.

Field compilation of reference data

Field work was conducted during the dry season between May and July 2013 with the identification of different cover types being guided by a field guide map that was prepared from unsupervised classification of a subset of the 2013 Landsat image which provided footprint coverage of the study area. Although the area has limited heterogeneity of natural vegetation types, the number of information classes in the field-guide map was purposefully set at 15 in order to facilitate reliable discretisation of the field-guide map into thematic classes that accommodated a wide range of other non-vegetation cover types and man-made features. Three sample sites were systematically identified to provide spatially representative coverage of all thematic classes. This procedure yielded 45 sample sites ($15 \times 3 = 45$) whose XY coordinates were displayed on the thematic

Table 1 Temporal sequencing and characteristics of images that were used

Satellite images	Scene ID	Path	Row	Acquisition date	Spatial resolution	Cloud cover	quality
Landsat 8 OLI	LC817108320 13212LGN01	171	083	31/06/2013	30 m	0.0%	9
Landsat 7 ETM+	LE717208320 03136ASN00	171	083	09/05/2003	30 m	0.0%	9
Landsat 5 TM	LT5170083198 4174XXX02	171	083	24/07/1993	30 m	0.0%	9
Landsat 5 TM	LT5171083199 3205JSA00	171	083	22/06/1984	30 m	0.0%	9
Aerial photo-mosaics							
True colour	ɸ	ɸ	ɸ	0 6/05/2013	0.5 m	*	*
Panchromatic	ɸ	ɸ	ɸ	27/06/2003	0.5 m	*	*
Panchromatic	ɸ	ɸ	ɸ	05/07/1994	0.5 m	*	*
Panchromatic	ɸ	ɸ	ɸ	17/05/1980	0.5 m	*	*

ɸ Details cannot be conveniently shown because photographs used were numbered differently according to flight paths. Image quality as rated by the USGS: 9 = excellent, 7–8 = good, 5–6 = fair, 1–2 = extremely poor; *Dry season aerial photographs acquired on cloud-free days

Sources: https://earthexplorer.usgs.gov.za; https://landlook.usgs.gov/viewer.html; DRDLR map by interactively using the enquire curser. These coordinates were then captured in a matrix table that summarised the spatial distributions of all sites.

Thereafter, all points were chronologically numbered from 1 to 45 and a detailed road map of the area overlaid on the thematic map to facilitate the identification of routes that were going to be used to access each of the sites targeted for investigation during fieldwork. To facilitate convenient navigation to all sample sites during fieldwork, the thematic map compiled at this stage was further segmented into 15 subsets each of which contained at least 3 sample sites. The identified sites in each of the 15 subsets were accessed with the aid of a Garmin GPS with a rated absolute positional accuracy of ± 4 m and investigated one after the other over a period of 15 intermittently distributed days. During field investigation, a geo-located hand-held camera photograph (photo-standard) was acquired for each sample site and a detailed description of what was observed recorded in a spread sheet. After detailed characterisation of cover types in all sample sites, the compiled field data were captured in a database file in which numbered-class-labelled sites and photo-standards were relationally linked to their coordinate locations. Successfully used by other researchers elsewhere (Campbell and Browder 1995), these photo-standards provided a reliable means of capturing spatial distributions of different cover types that were observed during field investigation.

Two-thirds of the information that was compiled during field investigation was summarised to produce a classification key with 9 cover types comprising (1) estuarine water, (2) bare area, (3) salt works, (4) salt marsh, (5) beach sand, (6) built-up areas, (7) *Zostera capensis*, (8) *Spartina maritima* and (9) terrestrial vegetation. Although different woody and herbaceous species were identified in the dryland peripheries of the salt marsh during field investigation, it was not possible to assign species-specific classes to this vegetation because of the nested distributions of different woody species and similar distributions of herbaceous species in a heterogeneous matrix. This limitation explains why species-level classes were not used to segregate different dryland vegetation species that were broadly classified as terrestrial vegetation. This aggregation was reasoned to be appropriate for this study because it accommodated the inability of Landsat imagery's coarse 30-m spatial resolution to discriminate sub-pixel sized cover types consisting of different vegetation species without compromising the main objective of the study.

Training area selection and signature evaluation

To compile signatures that are adequately representative of each class, a sampling between 10*n* and 100*n* pixels per class is recommended, with *n* being the number of bands used in the classification (Andersen 1998). Because 3 bands were used in this classification (bands 4, 3 and 2), the minimum number of pixels that was required to support the extraction of representative signatures for each class was $10 \times 3 = 30$. In addition to this, it is also recommended that more accurate signatures can be obtained by using several smaller homogenous samples than fewer larger samples (Swain and Davis 1978; Campbell 1981). We closely followed these recommendations and decided to use smaller 2×2 pixel windows to extract signatures from each training site and calculated that we needed to extract signatures from a minimum of 7.5 pixels (30 pixels/4 pixels in each window) which translated to approximately 3 training sites for every class. Following these calculations, we purposefully opted to use a higher threshold than the recommended minimum by extracting signatures from a minimum of 5 training sites for each class. In order to accomplish this, we complimented signatures extracted from 2/3 of the field sites we had targeted to use for supervised classification with additional signatures from training sites that were identified with the aid of the aerial-photo mosaics in our database. This procedure was accomplished by concurrently displaying each Landsat image and its corresponding aerial-photo mosaic in geographically linked viewers and using the Linked Cursor tool to confidently identify different cover types in the aerial photo-mosaics under appropriate magnification. Image correspondence was established by pairing Landsat images and aerial photo mosaics in the same order they are listed in Table 1 in order enhance close temporal correspondence.

Confident selection of additional training samples for dynamic vegetation cover types was enhanced by using ancillary information on preferred habitats by individual species and contextual information. Because *Zostera capensis* is known to occur as an intertidal species that flourishes below sea level to elevations of 0.9 m above sea level because of its tolerance of strong tidal conditions and periodic desiccation characteristics of the intertidal zone (Adams 2016) while *Spartina maritima* is largely a salt marsh plant (Leandro 2015), these species were expected to occur in the tidal and intertidal marshes and not in the dryland fringes of the estuary where thicket vegetation is dominant. Non-vegetation cover types that include beach sand and built-up areas were likewise confidently identified on the basis of contextual detectors that include appearance, tone, texture, shape, relative location and arrangement and orientation as suggested by Gurney and Townshend (1983). Overall, 4 class-labelled signature files were compiled and signature evaluation accomplished by using Euclidean distance to determine spectral separability in ERDAS IMAGINE. In performing this analysis, two pairs of signatures were used to determine the best minimum separability with 9 being set as the best minimum distance between 2 pairs of signatures. All cover types were spectrally separable because their separability values exceeded the minimum separability value.

Supervised classification and classification accuracy assessment

After ascertaining the separability of all classes, signature files that were compiled during the selection of training areas were used with the Mahalanobis distance classifier distance classifier to classify all images partly because it produced the best results compared with the maximum likelihood and minimum distance classifiers and also because it gives complete information classes without any unclassified pixels (Lillesand and Kiefer 2000; Tottrup and Rasmussen 2004; Andersen 1998). Class labels were assigned to output thematic maps on the basis of the nine class names that were assigned to individual sample sites during field investigation. Classification accuracy assessment was performed by using 1/3 of the field data reserved for this purpose during supervised classification and collateral information from aerial photographs. The steps that were involved in classification accuracy assessment comprised compilation of confusion matrixes for each of the four map outputs and calculation of producer, user and global accuracies and kappa coefficients (K) following procedures suggested by Campbell (2002). Accuracy levels for the 2013 Landsat 8 OLI, 2003 Landsat ETM+ and 1993 and 1984 Landsat 5 TM map outputs were (2013) 82.5%, $K = 0.806$; 77.5%, $K = 0.827$; 67.5%, $K = 0.782$ and 80%, $K = 0.825$, respectively.

Although these levels of accuracies are low by standards recommended by others (Thomlinson et al. 1999; Rogan et al. 2003), the notion of a universal cut-off point above which a classification can be considered to be accurate is no longer tenable. Hamandawana and Chanda (2010) provide an informative review of authoritative sources (Foody 2008; Trodd 1995) which demonstrates that the 85% used by many is a misapplication of pioneer work by Anderson et al. (1976) who suggested this as the target for mapping a small number (~ 9) of broad land-cover classes from coarse resolution (80 m) Landsat multispectral (MSS) imagery. Although a lot of factors can be invoked to explain why our accuracies were below the 85% cut-off point (Foody 2002), it has to be pointed out that levels of classification accuracy are meant 'to enable users to determine a map's suitability for their specific needs and not to provide a basis for quality assessment (Foody 2008) because map accuracies are not necessarily a true reflection of closeness to reality (Congalton and Green 1993)'. Likewise, it is also important to note that although the kappa statistic is widely used to classify accuracy, it tends to underestimate accuracies by removing chance agreement from the quantification process (Foody 2008). In view of these considerations, it is not unreasonable to conclude that levels of accuracy of our map outputs were within acceptable limits.

Presentation of results and statistical analysis

Presentation of results

Results of this investigation are presented in the form of a table (Table 2) that shows percentage changes in information classes that were mapped and data list names that were used to describe them in Excel and a graph (Fig. 2) that shows temporal variations in the spatial distributions of these cover types.

All cover types exhibited marginal changes, with bare area, salt-works and terrestrial vegetation decreasing by no more than approximately 7% for each of these cover types while each of the remaining cover types increased by less than 4%. The long-term increase in *Zostera capensis*, *Spartima maritima* by 2.39%, closely mimicked the observed increase in estuarine water by 2.17% while salt marsh marginally increased by 0.22% with this increase being punctuated by initial decrease and terminal increase (Table 2).

Estuarine water and *Spartima maritima* exhibited disruptive trends characterised by abrupt increases in 2003 while bare area, salt works and terrestrial vegetation persistently decreased as bare sand, built-up area

Table 2 Percentage changes in information classes that were mapped: 1984–2013

Cover type	Data list name	Percentage composition				Percentage change			
		1984	1993	2003	2013	1984–1993	1993–2003	2003–2013	1984–2013
Bare area	Barea	11.5	9.6	8.7	7.4	− 1.9	− 0.9	− 1.3	− 4.13
Salt works	Sawks	10.9	9.3	5.7	3.9	− 1.6	− 3.6	− 1.8	− 6.96
Salt marsh	Samas	18.3	18.0	17.8	18.5	− 0.3	− 0.2	0.7	0.22
Beach sand	Bsand	6.5	7.0	8.0	10.2	0.5	1.0	2.2	3.70
Built-up areas	Buarea	4.3	5.9	6.5	7.0	1.6	0.6	0.5	2.61
Estuarine water	Estwat	7.0	8.9	10.4	9.1	1.9	1.5	− 1.3	2.17
Spartina maritima	Spama	16.7	17.4	19.3	19.1	0.7	1.9	− 0.2	2.39
Zostera capensis	Zocaps	10.0	10.0	10.2	12.4	0.0	0.2	2.2	2.39
Terrestrial vegetation	Tereveg	14.8	13.9	13.3	12.4	− 0.9	− 0.6	− 0.9	− 2.41
Total area mapped in ha		460	460	460	460	–	–	–	–

and *Zostera capensis* progressively increased (Fig. 2). Although the graph is helpful by providing a snapshot overview of the direction of observed changes in different cover types, it has limitations because unlike statistical analysis, visualisation alone cannot adequately quantify the magnitude of a trend as people tend to focus on outliers so that strong variation can mask trends while gradual changes are difficult to detect from visual inspection. This limitation was overcome by objectively determining the direction of change for each cover type through statistical analysis.

Statistical analysis of results

The direction of observed changes in different cover types was determined by computing trend coefficients in Microsoft Excel which were double-checked by performing the Mann–Kendall (M–K) test to calculate the Sen Slope Estimate (SSE) for each cover type with statistical significance being determined by calculating p values at σ 0.05. The M–K test is a non-parametric test for identifying trends in time series data by comparing the relative magnitudes of sample data rather than the data values themselves (Gilbert 1987) while the SSE provides objective estimates of the magnitude of change (Sen 1968). A positive (negative) SSE indicates an upward (downward) trend while its magnitude indicates steepness (Gocic and Trajkovic 2013; Pandit 2016). The main advantages of the M–K test are that apart from being able to show whether a trend has been stationary, decreasing or increasing (Bryhn and Dimberg 2011), it does not require that the data should conform to any particular type of distribution (https://vsp.pnnl.gov/help/vsample/design_trend_mann_kendall.htm;

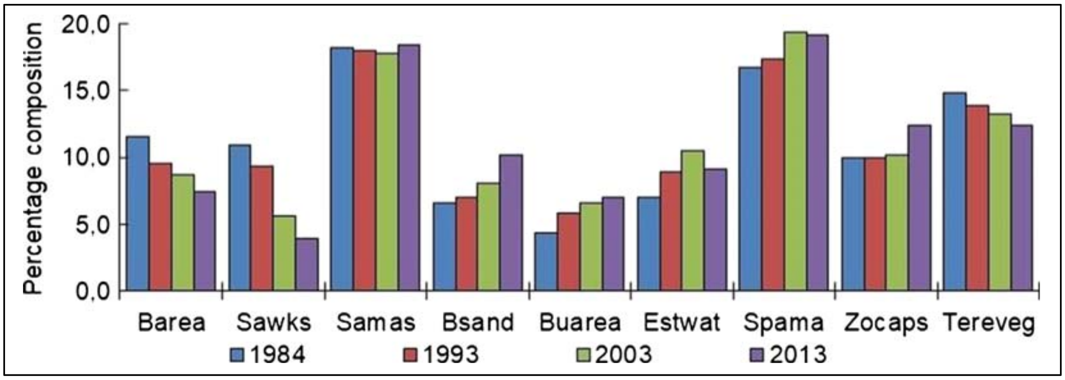

Fig. 2 Temporal variations in spatial distributions cover types that were mapped: 1984–2013

Meals et al. 2011) and it is not sensitive to abrupt breaks in datasets due to inhomogeneous time series (Tabari et al. 2011; Jaagus 2006; Isioma et al. 2018). Table 3 summarises results of the statistical analyses that were performed as described above.

Simple linear trend analysis revealed negative trends for bare area, salt works and terrestrial vegetation and positive trends for salt marsh, beach sand, built-up areas, estuarine water, *Zostera capensis* and *Spartina maritima* while the SSE revealed slightly different results in which salt marsh exhibited a false positive trend.

Discussion

Results of this investigation point to marginal long-term changes in most cover types below ± 3% the only exceptions being beach sand which increased by 3.7% and bare area and salt works which declined by 4.13% and 6.96% respectively (Table 2). Although changes in all cover types were not statistically significant at σ = 0.05, the persistent expansion of built-up areas by 2.61% and inversely related decrease in terrestrial vegetation by 2.41% provide a convenient entry point for interrogating the major drivers of changes in other cover types. Human agency emerges as the most likely cause of the observed decrease in terrestrial vegetation which was largely induced by conversion of vegetated dryland fringes of the estuary for residential and other development activities with the same conversion also explaining the decrease in bare area by 4.13%. Although it is generally recognised that there is need to regulate urban activities and control the removal of vegetation on river banks and fast track the obliteration of redundant structures that include decommissioned salt-works and quarries, the major reported constraints that have made this unattainable include inadequate capacities to educate local communities, lack of incentives to motivate voluntary participation and the high costs associated with restorative interventions (ZRWRMP 1999). These changes and challenges have nested implications for the natural composition of ecosystems in this environment with loss of biodiversity emerging as one of the adverse effects of human interference.

Land conversion unavoidably induced loss of habitat for salt-tolerant dryland species which is likely to have been fast-forwarded and aggravated by sea level rise induced loss of terrestrial habitat and reduction in biodiversity. These adverse effects are corroborated by Bornman et al. (2016) who report considerable development-induced (housing, roads, railways and industry) alteration of ~ 118 ha of the terrestrial and ecotone areas adjacent to the estuary and an estimated loss of ~ 98 ha of sandbank and relative sea level rise (RSLR) induced inundation of ~ 41 ha between 1939 and 2012. The last phenomenon is corroborated by Kristensen (2004) who reports that in South Africa, climate change is one of the main driving forces affecting coastal water ecosystems. This observation is indeed consistent with findings of our investigation that indicate a long-term 2.17% expansion of estuarine water between 1984 and 2013 (Table 2) with continuous discharge of treated sewage into the Swartkops River via the Kat and Motherwell Canals amplifying this natural increase by inducing, higher than natural flow in the lower reaches of the river (Steward et al. 2010; Lord

Table 3 Linear trend coefficients, Sen Slope Estimates and *p* values for observed changes in cover types that were mapped: 1984–2013

Cover type	Linear trend coefficient	R^2	SSE	*p* value
Bare area	$y = -0.1355x + 280.12$	$R^2 = 0.9715$	− 0.1357	*0.0833
Salt works	$y = -0.2557x + 518.4$	$R^2 = 1.0000$	− 0.2557	*0.0833
Salt marsh	$y = 0.0046x + 9.0236$ (Ᵽ)	$R^2 = 0.0339$	− 0.0066	*0.7500
Beach sand	$y = 0.1255x - 242.84$	$R^2 = 0.9189$	0.1138	**0.0833
Built-up areas	$y = 0.089x - 171.83$	$R^2 = 0.9025$	0.0766	**0.0833
Estuarine water	$y = 0.0789x - 148.85$	$R^2 = 0.4978$	0.1112	**0.3333
Zostera capensis	$y = 0.0772x - 143.65$	$R^2 = 0.6829$	0.0514	**0.0710
Spartina maritima	$y = 0{,}0936x - 168{,}97$	$R^2 = 0{,}8443$	0.0839	**0.3333
Terrestrial vegetation	$y = -0.0809x + 175.18$	$R^2 = 1.0000$	− 0.0809	*0.0833

Interpretation: No trend if the *p* value is > 0.05; negative SSE = declining trend and vice versa

*Declining but not significant; **Increasing but not significant; (Ᵽ), false positive linear trend coefficient et al. 1991) and prolific growth of water hyacinth (*Eichhornia crassipes*) in the upper reaches of the estuary (ZRWRMP 1999) which needs to be eliminated by undertaking regular clean-up efforts that have been successfully implemented since then up to the present to eradicate this aggressive encroacher (http://www.sabcnews.com/sabcnews/swartkops-river-in-e-c-gets-a-clean-up/). Although some authors are of the view that most salt marsh estuaries can keep pace with RSLR by migrating upstream (Kirwan et al. 2016), this adjustment can be muted by river mouth characteristics, i.e. confinement of estuaries to drowned river corridors, sills, sand banks, channel width and depth (Schumann 2013) and non-natural barriers in upstream areas (roads, artificial embankments, railway lines, bridges, stabilised banks, in-channel and upstream dams, housing developments, industrial areas etc.) that mediate and prohibit the extension of estuaries into upland areas (Bornman et al. 2016).

Although RSLR should have induced a corresponding decrease in beach sand, results of our investigation indicate the opposite, with this cover type increasing by 3.7% between 1984 and 2013. This counterintuitive scenario can be explained by significant reduction in sediment erosion after construction of the Settlers Bridge (Fig. 1a) which constrained big floods to only one area of the estuary's terminal reaches (ZRWRMP 1999) due to induced obliqueness to the inlet channel's natural parallel orientation to the coast (Esterhuysen and Rust 1987; Fig. 1a) and reduced flashing of sediments into the sea due to impoundment of channel flow in upstream dams. It is however worth noting that the earlier stated loss of ~98 ha of sandbank reported by Bornman et al. (2016) is not necessarily a contradiction of our observed increase in beach sand because this is likely to be indicative of observational dissimilarities arising from differences in footprint coverages that were used to delimit the spatial extent of the SRE. We consistently mapped the same but smaller area covering 460 ha (Table 2) compared with the variable areas (602.33 ha and 695.25 ha excluding development) that were mapped by Bornman and co-authors for 1939 and 2012 respectively, with area differences between the latter 2 being explained by lack of images that provided footprint coverages at both time slices.

In the tidal, intertidal and supratidal environment, the most dominant vegetation species that were mappable as individual cover types from Landsat imagery were *Spartina maritima* and Zostera *capensis*. Long-term changes in these cover types were in the same direction, with *Spartina maritima* and *Zostera capensis* increasing by 2.39% (Table 2). The long-term expansion of these cover types by the same percentage is interesting because it suggests the influence of tolerance to similar habitat conditions and external stressors. The ecological importance of these species to salt marsh ecosystem processes has been reviewed by many (Phair 2016; James and Harrison 2010; Whitfield and Cowley 2010; Siebert and Branch 2006; Beckley 1983). They play a vital role by providing spawning grounds and nurseries for marine fish species and provide habitats for estuarine-resident species and recycle nutrients and help to maintain the trophic functioning and productivity of shallow waters by providing food to large and small herbivores and help to control erosion by sea waves and river-water. These and other functions not considered here demonstrate the importance of these species and why they should be conserved. Fortunately, results of our investigation suggest that in this environment, these species are enjoying hospitable habitat conditions that do not threaten their sustainability. Factors that provide plausible explanation of the observed increase in these species include the sustained increase in estuarine water (Table 2 and Fig. 2) which provided more habitat and nutrient enrichment by storm water runoff and sewage water which is regularly discharged into the Swartkops River (Bornman et al. 2016; Nel 2014; Steward et al. 2010; Knox 2003; Lord et al. 1991). Although nutrient enrichment from planned sewage disposal was evidently beneficial for these species, this is not to suggest that this practice should be encouraged because excessive nutrient loading can have detrimental feedback effects that can adversely impact on these species by promoting eutrophication, algal growth and competition for light and nutrients (Hemminga and Duarte 2000).

While most of the cover types that were mapped changed as described above, the most and least pronounced changes were for salt works and the salt marsh which declined and increased by 6.96% and 0.22%, respectively (Table 2). The former consist of salt production facilities operated by Cerebos Industries and Marina Sea Salt by using river water abstracted into evaporation pans that were purposefully located in upland fringes at elevations above the water level in the Swartkops River. Persistent decrease in the area covered by these pans from 1984 up to 2013 is explained by phased decommissioning in successive stages as increased efficiency enabled the industries to maintain and increase production. Because of high concentrations of residual salt in decommissioned pans, restorative rehabilitation of disused sites has been problematic and they have tended to persist as man-made seasonal salt pans. The marginal increase in salt marsh by 0.22% is interesting because it points to contradictory scenarios that can suggest stable variation or substantial change depending on procedures used for time-series trend analysis. Although simple linear trend analysis yielded a positive trend for this cover type, the SSE that was used to crosscheck the reliability of the former technique revealed a long-term decrease (Table 3). The take-home message from these discordant results is that simplistic analysis of disruptive changes above and below a small margin can give a misleading sense of stability by failing to capture subtle inter-cover variations in the magnitude of change over long time periods (Fig. 3).

The variations in the magnitude and direction of change summarised in Fig. 3 provide additional insights that can be easily missed by depending on simplistic statistical analysis. Although changes for all cover types were marginal and variable, the positive change captured for saltmarsh by simple linear trend analysis was actually indicative of a long-term decrease in this cover type because the isolated occurrence of the only observed increase in 2013 (Table 2 and Fig. 2) skewed the long-term trend in a positive direction. It is also apparent that the greatest magnitude of change was for salt works (SSE = − 0.2557, $R^2 = 1.00$) while terrestrial vegetation had a lower SSE value of − 0.0809 even though its R^2 value was the same as that for salt works (Table 3). The discordant SSE values for these cover types provide informative insights by indicating that salt works declined at a faster rate than terrestrial vegetation which, the R^2 value failed to capture. What this suggests is that it is necessary to explore different techniques in order to untangle the complexity of complex ecosystems. With the aid of these additional insights, management interventions can easily be directed toward those cover types where changes are large enough to warrant timely or immediate attention. If one proceeds to further examine the changes summarised in Fig. 3 as described above, it becomes immediately apparent that development-driven decrease in terrestrial vegetation, for example, is a major cause of concern while changes in salt marsh may be judged to be within tolerable limits if the primary management objective is to enhance timely containment of persistent loss of biodiversity. This kind of analysis can be used to enhance the articulation of objectively informed management options by extending it to include the remaining cover types.

Conclusion

The objective of this paper was to provide a multi-temporal reconstruction of spatial changes in land cover in and around the SRE. We did this by using multi-date remotely Landsat images to map and quantify temporal variations in 9 cover types comprising (1) estuarine water, (2) bare area, (3) salt works, (4) salt marsh, (5) beach sand, (6) built-up areas, (7) *Zostera capensis*, (8) *Spartina maritima* and (9) terrestrial vegetation. In general, results of our investigation strongly suggest that the SRE is largely affected by two main drivers, natural (mainly climate) and anthropogenic (e.g. development) which cause wide-ranging pressures on the entire ecosystem that result in impacts on freshwater resources, either by changing the quantity of ground and/or surface

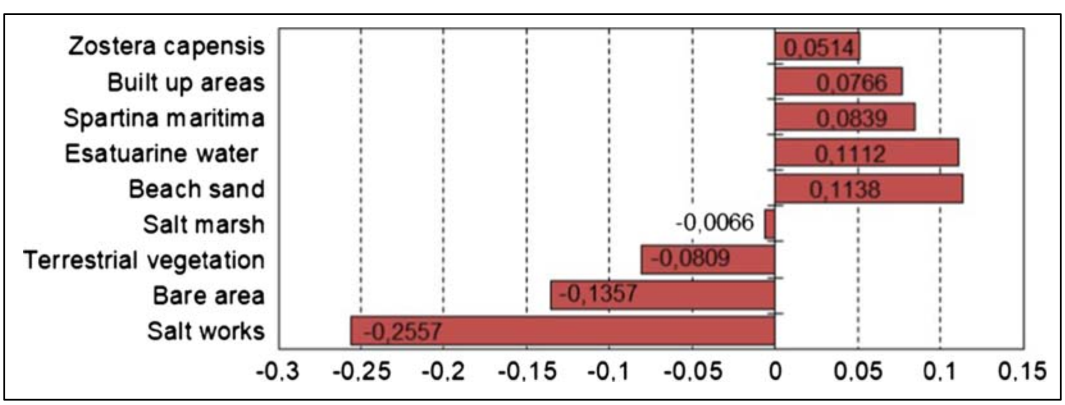

Fig. 3 Sen Slope Estimates for cover types that were computed from the M–K test

water, and/or by changing the quality of ground and/or surface water. Specifically, the findings of our investigation point to *human-induced loss of biodiversity* due to persistent encroachment of different activities on terrestrial vegetation, *substantial expansion of the salt marsh* due to the combined effects of climate change–driven relative sea level rise and planned discharge of sewage water and storm water runoff from impervious built-up areas, and *persistent increase in keystone salt marsh vegetation species* notably *Zostera capensis* and *Spartina maritima* due to the combined influence of human-induced nutrient loading into estuarine water and relative sea level rise. These observations argue for the immediate need to embrace appropriately informed management interventions in order to enhance the sustainability of salt marsh ecosystems for the benefit of present and future generations.

Accomplishing this requires (1) constant monitoring and compilation of robust datasets that can be used to provide reliable information, (2) capturing this information in a framework that enables it to be easily distributed for research, planning and management by enhancing the adoption of objectively informed interventions, (3) embracing and enforcing regulations that discourage/prohibit direct and indirect disposal and discharge of waste and pollutants into estuaries, (4) exploring climate-friendly water-use strategies that can be used to reduce abstraction of stream and groundwater flows into estuaries by enforcing regulations that reduce entitlements and (5) increasing water use efficiency and the exploitation of limited water supplies through rainwater harvesting, efficient irrigation techniques in catchment areas and non-wasteful domestic and industrial water use practices, (6) avoiding surface and groundwater extraction and abstraction beyond recharge levels, (7) land use zoning in order to control the encroachment of environmentally unfriendly land use practices into the estuarine environment and its catchment areas, (8) planned regulation of consumptive and recreational uses of estuaries, (9) sensitizing and incentivising local communities to embrace responsible stewardship of resources within their environment, (10) forward planning to identify and implement interventions that are potentially capable of mitigating the adverse effects of climate change–driven relative sea level rise and changes in habitat conditions for wide-ranging aquatic and terrestrial species and (11) holistic conservation approaches that recognise estuaries as part of the broader ecosystems in which they are situated. We therefore urge planners, policy makers and other interested stakeholders to give priority consideration to the identification and timely implementation of sustainable resource use practices and invite those interested to complement this initiative by making concerted efforts to constantly monitor salt marsh ecosystems in order to enhance their sustainability for the benefit of present and future generations.

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