BEFORE THE NATIONAL GREEN TRIBUNAL (SZ) CHENNAI

MEMORANDUM OF APPLICATION
(Under Section 18(1) read with Sections 16 (h) of National Green Tribunal Act, 2010)

Appeal No.14 of 2022

M. Yuvadeeban ...Appellant

Vs.

Department of Fisheries & Ors. ...Respondents

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FILE – H

Through
Yogeshwaran. A
Counsel for Appellant

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(Under Section 18(1) read with Sections 16 (h) of National Green Tribunal Act, 2010)
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M. Yuvadeeban ...Appellant

Vs.

Department of Fisheries & 2 Ors. ...Respondents

AFFIDAVIT DATED 15.03.2023 FILED BY THE APPELLANT

I, M. Yuvadeeban s/o Margaret Lawrence, aged about 26 years, residing at B2, Ramaniyam Marvel, Seshadripuram, 1st main road, Velacherry, Chennai 42, do hereby solemnly affirm and sincerely state as follows:

1. I am the Appellant herein and am aware of the facts and circumstances of the case and am competent to affirm to the contents of this affidavit.

2. I submit that during the course of arguments, the project proponent submitted that there was nothing called a comprehensive EIA and that their EIA, allegedly based on 3-month data i.e a rapid EIA was sufficient. It was also submitted that the references to comprehensive EIA in the ToR should be read as Detailed EIA.

3. In this context, I am advised to submit that the project proponent has failed to consider clause 4.2 of the CRZ Notification 2011 – which mandates comprehensive EIAs for projects located on medium and low erosion stretches. Under clause 3 (viii) of the Notification, ports and harbors are prohibited in high eroding stretches. In low and medium eroding stretches, such projects require a comprehensive EIA.

4. It is submitted that Rapid EIA and Comprehensive EIA differ in the time scale of the data supplied. Rapid EIA is based on one season data collection and

M. Yuvaraj

comprehensive EIA is based on at least 3 season data. Since the project proponent denied that there was nothing called a comprehensive EIA, it is necessary to submit the response of the then Environment Minister in the Lok Sabha on 5.05.2015 to UNSTARRED QUESTION NO: 6221 for the consideration of this Hon'ble Tribunal and the EIA Manual of the MoEF&CC available at https://moef.gov.in/en/eia-manual/ - the Introduction chapter of the EIA manual is also produced herewith.

5. It is submitted further, the project proponent argued that there exist only "seagrass patches" and not seagrass beds in the estuary. Firstly, the presence of seagrass species in the estuary has been admitted in the report of the Annamalai university submitted by the project proponent. Secondly, there are copious references in literature to the presence of seagrass beds in the estuary. The terminology "patches" sought to be used makes no difference – it is a fact that the shallow areas of the estuary, especially the areas nearer to the banks are covered with extensive seagrass beds. The CRZ Notification seeks to protect seagrass beds and classifies them as an ecologically sensitive area, protected as CRZ IA.

6. The appellant has already filed videos of seagrass beds at the Azhagankuppam project site where activity was started and gravel and debris has been dumped by the project proponent. These shores were visited again on 26.02.2023 and videos were made using a gopro camera as well as a mobile phone camera. The entire area, even along-side the area where construction materials have been dumped inside the water (which has been disturbed by the dumping of foreign material), are covered with extensive seagrass beds. Screenshots from the videos are pasted below and the following drive link has all the videos and photographs: https://drive.google.com/drive/folders/1e4C5Z5lBcqwL2OD45itLpJIC_s_QnYs?usp=shar

M. Yuvaraj

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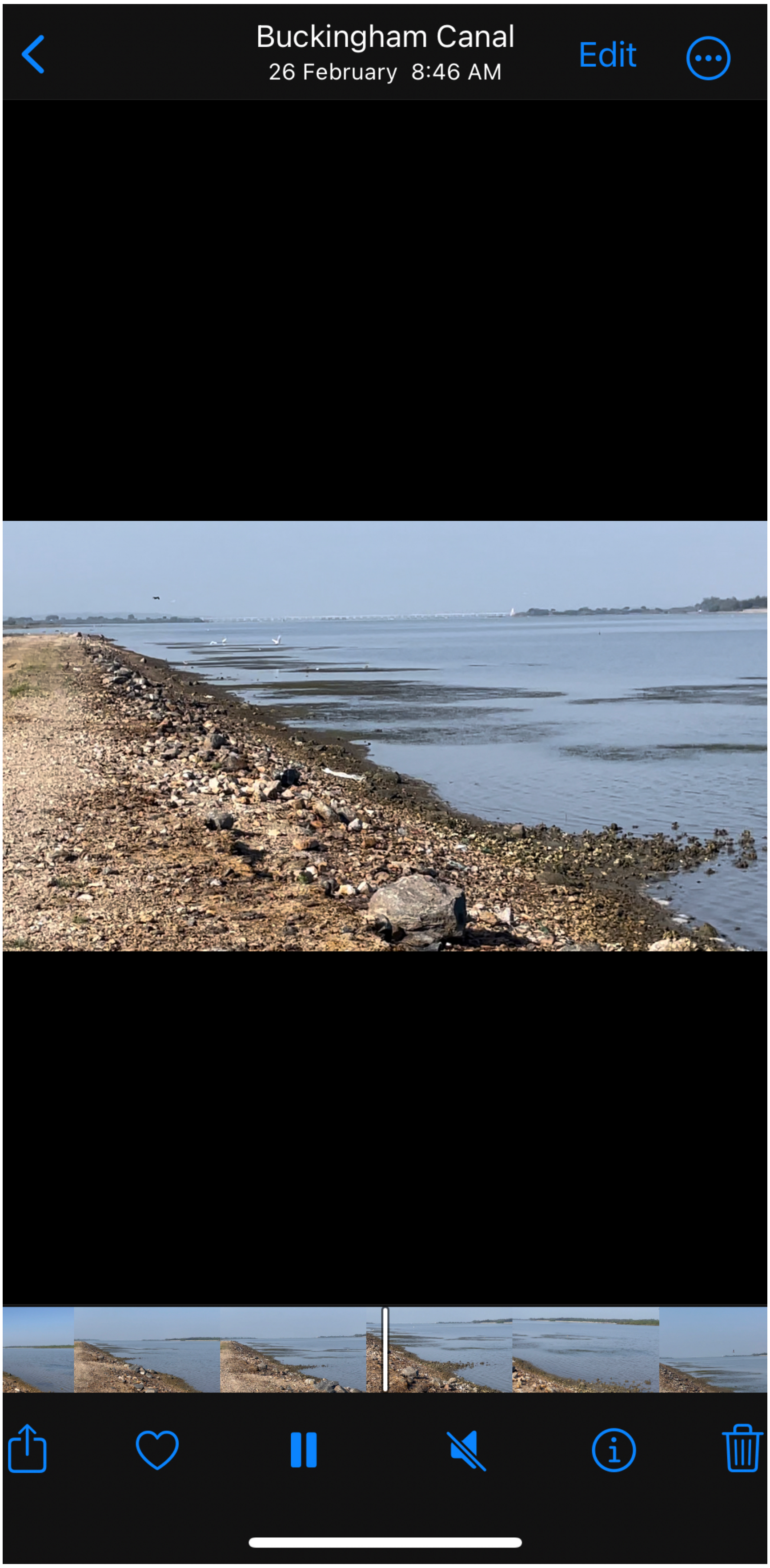
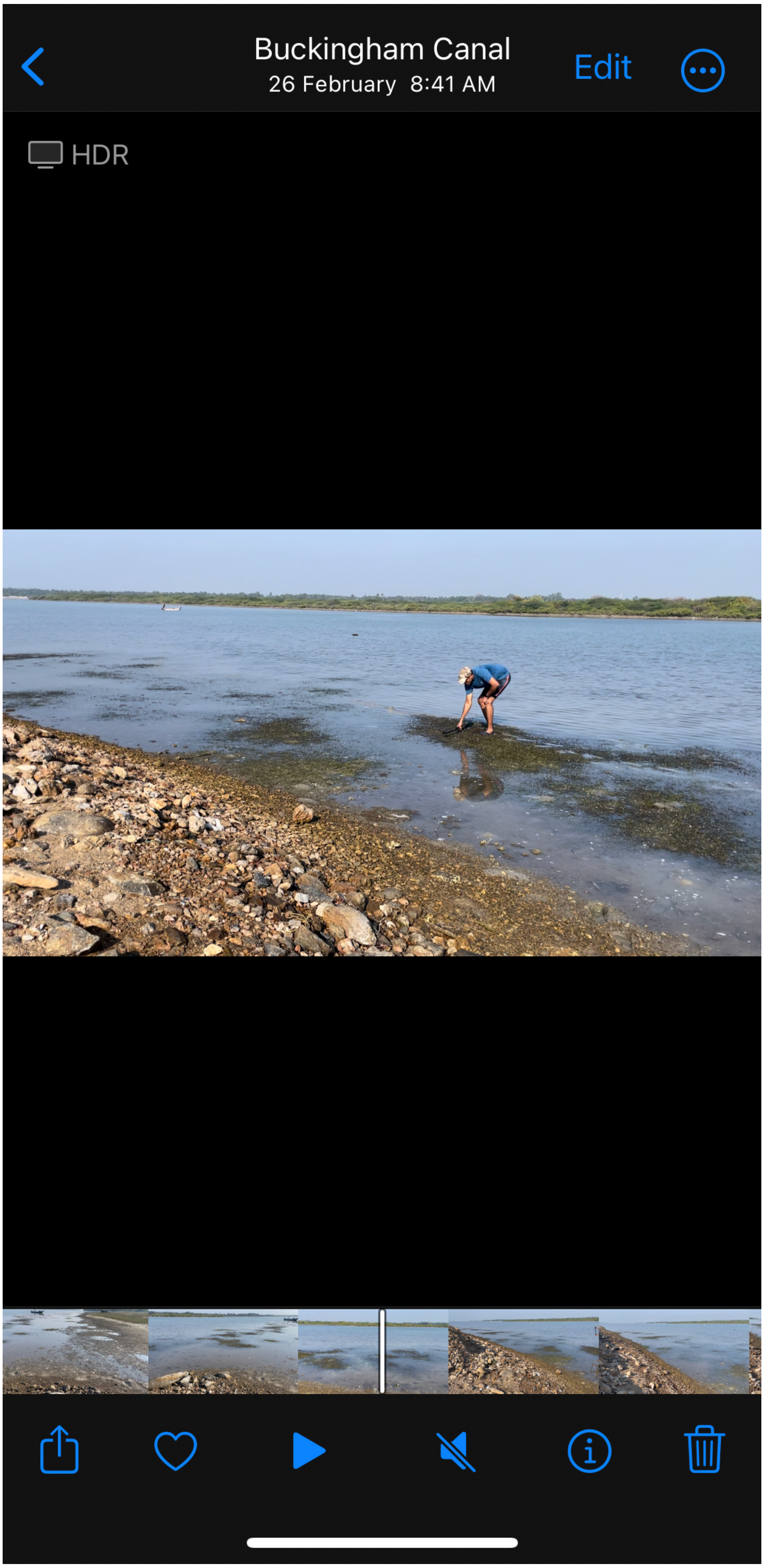
M. Yuvaraj



M. Yuvaraj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7. The harbor site at Alamparaikuppam , as submitted in the previous pleadings of the appellant is a salt marsh and a biologically active mudflat – the area is home to oyster beds, mangroves etc. The mudflats are teeming with biological activity – the videos showing girdled horn snail, fiddler crabs in the mudflats are also filed. These biologically active mudflats, mangroves are also protected under CRZ IA.

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8. It is submitted that sheltered marine spaces are required for the existence of sea grass, oyster reefs, mudflats and salt marshes. This is provided by buffers like sand bars at the estuary mouth which reduce wave velocity and flow rate. This in turn makes it conducive for shorebirds. Studies regarding the effect of dredging on sea grass beds and the ecological function of tidal flats and shallow water habitats are produced herewith.

9. It is submitted that studies recording erosion of the shoreline post the construction of the Dharma port are also produced herewith. These studies conclude that " the area of Dhamra coast was under accretion during 1990-2000 and this area experienced erosion near Dhamra port just after the development of the port in 2007. This study concludes that shoreline of study area is under high risk of erosion and inundation due to natural as well as anthropogenic activities in the area". Another study (Mohanty) also concludes that " although accretion pattern was witnessed along Dhamra port from 1990-2000, drastic erosion was observed after the port area development in 2007".

10. It is therefore prayed that this Hon'ble Tribunal be pleased to consider the above submissions and pass necessary orders as deemed fit proper and necessary in the facts and circumstances of the case.

M. Yuvaraj

Solemnly affirmed and signed his name this the 15th day of March, 2023 At Chennai

BEFORE ME

Chethana V
(MS-1108/18)
Chethana. V
No. 18, Brindhavan St.
Mylapore, ch-04

ADVOCATE : CHENNAI

EIA MANUAL

EIA Manual

ENVIRONMENTAL IMPACT ASSESSMENT

Impact Assessment Division

Ministry of Environment and Forests

Government of India

January, 2001

A Manual

Preface

Environmental Protection and Sustainable Development have been the cornerstones of the policies and procedures governing the industrial and other dev India.Ministry of Environment & Forests has taken several policy initiatives and enacted environmental and pollution control legislations to prevent indisc natural resources and to promote integration of environmental concerns in developmental projects. One such initiative is the Notification on Environmen (EIA) of developmental projects issued on 27.1.1994 under the provisions of Environment (Protection) Act, 1986 making EIA mandatory for 29 catego projects.One more item was added to the list in January, 2000.

EIA is a planning tool that is now generally accepted as an integral component of sound decision-making. The objective of EIA is to foresee and address p problems/concerns at an early stage of project planning and design.EIA/EMP should assist planners and government authorities in the decision making pro key impacts/issues and formulating mitigation measures.Ministry had issued sectoral guidelines some time ago.A compendium of the procedures and Application Form and Questionnaire for Environmental Clearance was published in September, 1999 in association with the Confederation of Indian Industry.

As part of the continued efforts to ensure transparency in the procedures of environmental clearance and to assist the project authorities in improving the qu this Manual is now being brought out by the Ministry.The Manual has been designed to cover the whole gamut of issues like regulatory requirements, the EIA baseline studies, identification of key issues and consideration of alternatives, impact analysis and remedial measures in a systematic way.It also del reviewing the adequacy of EIA and EMP reports and post-project monitoring.To make the Manual comprehensive and self-contained, information pertainin base line data generation and monitoring, thumb rules for pollution control measures etc. have been annexed to the main text.

It has been our experience that EIA documents are often voluminous but much of the base-line information included in these is not fully utilised in th prediction.Some of the impacts of the proposed development are of little significance to the decision making process. Hence a brief reference has been mac on Risk Assessment and Hazard analysis also has been included.

It is hoped that project proponents, EIA consultants and regulatory authorities will find this EIA Manual useful.

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Ministry of Environment, Forest and Climate Change
Indira Paryavaran Bhawan
Jorbagh Road New Delhi – 110 003
INDIA.

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ENVIRONMENT IMPACT ASSESSMENT

MINISTRY OF ENVIRONMENT, FORESTS AND CLIMATE

EIA GUIDELINES FOR PORTS AND HARBOURS

05th May 2015

LSQ *547

RAMSINH PATALYABHAI RATHWA

Will the Minister of **ENVIRONMENT, FORESTS AND CLIMATE CHANGE** be pleased to state:-

(a) the guidelines laid down for regulating the Environmental Impact Assessment (EIA) studies for the ports and harbours in the country including existing criteria for development projects;
(b) whether the Government has received representations from various States including Gujarat to accord clearance to projects in coastal stretches without comprehensive EIA report based on aforesaid guidelines; and
(c) if so, the details thereof along with the steps being taken by the Government in this regard?

MINISTER OF STATE (INDEPENDENT CHARGE) FOR ENVIRONMENT, FOREST AND CLIMATE CHANGE (SHRI PRAKASH JAVADEKAR)

(a) to (c) A statement is laid on the table of the House.

Statement referred to in reply to parts (a) to (c) of the Lok Sabha Starred Question No. 547 by Shri Ramsinh Rathwa regarding "EIA Guidelines for Ports and Harbours" for 05.05.2015:

(a) All developmental projects as specified in the EIA Notification, 2006 require prior environmental clearance. The prescribed guidelines for conduct of EIA study necessitate that sector and location specific Terms of Reference for the study is approved by the Expert Appraisal Committee constituted under EIA Notification, 2006.

(b) and (c) A request was received from the Government of Gujarat to consider port and harbour projects for Coastal Regulation Zone (CRZ) clearance in low and medium eroding coastal stretches based on rapid Environment Impact Assessment (EIA) instead of comprehensive EIA Report. A rapid EIA report is prepared based on one season data whereas the comprehensive EIA is prepared based on three season data. This requirement is uniformly applicable throughout the country. Hence, the proposal could not be considered.

****

EIA FOR LOW MEDIUM EROSION

05th May 2015

LSQ 6221

JAYSHREEBEN PATEL

Will the Minister of **ENVIRONMENT, FORESTS AND CLIMATE CHANGE** be pleased to state:-
(a) whether the State Government of Gujarat has requested the Union Government to make modification in the policy for low and medium erosion areas to have Rapid EIA instead of comprehensive EIA for getting environment and CRZ clearances which will result in early completion of new projects or expansion of projects;
(b) if so, the details thereof and the reaction of the Government thereto; and
(c) the time by which the proposal is likely to be finalized?

MINISTER OF STATE (INDEPENDENT CHARGE) FOR ENVIRONMENT, FOREST AND CLIMATE CHANGE (SHRI PRAKASH JAVADEKAR)

(a) to (c) The Government of Gujarat requested to consider projects for Coastal Regulation Zone (CRZ) clearance in low and medium eroding coastal stretches based on rapid Environment Impact Assessment (EIA) instead of comprehensive EIA Report. The request could not be considered since a rapid EIA, which is based on one season data may not address all the environmental concerns. As per the procedure prescribed for seeking prior clearance under the CRZ Notification, 2011, all project proposals in stretches classified as low and medium eroding as well as stable coasts shall be accompanied by comprehensive EIA studies based on three season data. This requirement is uniformly applicable throughout the country. To conserve and protect such coastal stretches, promote development through sustainable manner, it is important that comprehensive EIA studies are carried out based on scientific principles and Environment Management Plans worked out accordingly before considering proposals in such stretches.

MINISTRY OF ATOMIC ENERGY CHANGE

EIA REPORT ON MITHIVIRDI NUCLEAR POWER PLANT

17th December 2014

LSQ 3955

KIRIT PREMJIBHAI SOLANKI

Will the Minister of ATOMIC ENERGY be pleased to state:-
(a) the main findings of the Environmental Impact Assessment (EIA) of the proposed Mithivirdi Nuclear Power Plant in Gujarat;
(b) whether the Government has taken any measures to address the issues flagged in the report; and
(c) if so, the details thereof and if not, the reasons therefore?

THE MINISTER OF STATE FOR PERSONNEL, PUBLIC GRIEVANCES & PENSIONS AND PRIME MINISTER'S OFFICE (DR.JITENDRA SINGH)

(a) The Environmental Impact Assessment (EIA) report in respect of the proposed nuclear power plant at Chhaya Mithi Virdi in Gujarat has found that the setting up of the nuclear power plant at the site would not adversely affect the environment around the site.

(b)&(c) The application of Nuclear Power Corporation of India Limited (NPCIL) for environmental clearance including the EIA report is being reviewed by the Expert Appraisal Committee (EAC) of the Ministry of Environment & Forests (MoEF). The EAC during its meeting held on May 06, 2014 had sought additional information / details, which have since been submitted to the EAC by NPCIL.

Annex VI

Checklist for Ecological Impact Assessment

While verifying the Impacts on ecology delineated in the Impact Assessment statement, the reviewer may consider such of the following matters that are relevant to the proposed development:

- The general character of the existing site in terms of fauna and flora; landscape and geological features, lakes, creeks, marsh, mangroves, coral, forest and bush, sand dunes, mud flats, breeding and spawning grounds, habitats, flight paths, migratory paths and aesthetics.
- The consistency of the proposed development with any relevant statutory instruments, planning policies, heritage orders, measures under tribal or native people legislation, or international conventions (protecting, say, wetlands and migratory birds, or threatened or endangered species).
- Alternative sites for the proposed development, or alternative designs or techniques, which might pose reduced ecological risks. Reasons why this site is clearly preferable to all others.
- In that event, an ecological inventory of at least the most endemic and endangered species with major plant and animal habitats, particularly habitats critical to the preservation of threatened or endangered species. The geographical relationship of species on the site.
- Artificial features of the site as existing, such as roads, railways, buildings and other facilities relating current uses to the local ecology: agricultural activities.
- A history of tribal activity on the site, with reference to archaeological, cultural, and heritage items.
- Outstanding individuals such as the oldest or largest of the trees; rare or uncommon species, races, variants, and populations; unique or scarce habitats. Communities threatened or endangered.

- Plants or animals that could affect public health or safety: allergenic plants, poisonous and venomous species, pest or nuisance population; populations that might expand dramatically if the immediate environment were changed.
- The possible effects of the proposed development on terrestrial species (plants and animals); on aquatic species (fauna, fish, coral); on habitats; on the aesthetics of the site; on natural resources such as soil, geological formations, dunes, beaches, lakes, forest (including rain forest), coral reefs, mangroves, swamps, outcrops, and the atmosphere; including the possible effects of noise.
- The implications of clear felling or selective logging for timber and other forest products; the effects of road-building, drainage of wet areas, and the skidding, hauling of logs; the possibility of replacement by mono culture plantations; the danger of forest fragmentation causing genetic isolation of animal populations.
- The possibility of upsetting the species composition by excessive harvesting of fish, molluscs, crustaceans, seaweed, and other creatures and organisms.
- The possibility of the mining of coral for cement, lime, road-building and construction purposes; and other damage to coral.
- The threat to mangroves from clearing and development, and from pollutants.
- Other related developments in the area, which might have a cumulative ecological impact.
- Primary and secondary impacts, temporary and long-term, unavoidable impacts and risks; synergism; trans boundary effects; possible irreversible changes.
- The possible mitigation of effects by technical, or financial measures, by redesigning.
- Proposed post project monitoring.
- In sum the ecological significance of the site for the community and the potential for genuine loss due to the project.

GOVERNMENT OF INDIA

MINISTRY OF ENVIRONMENT, FORESTS AND CLIMATE CHANGE

LOK SABHA

UNSTARRED QUESTION NO: 6221

ANSWERED ON:05.05.2015

EIA FOR LOW MEDIUM EROSION

JAYSHREEBEN PATEL

(a) whether the State Government of Gujarat has requested the Union Government to make modification in the policy for low and medium erosion areas to have Rapid EIA instead of comprehensive EIA for getting environment and CRZ clearances which will result in early completion of new projects or expansion of projects;

(b) if so, the details thereof and the reaction of the Government thereto; and

(c) the time by which the proposal is likely to be finalized?

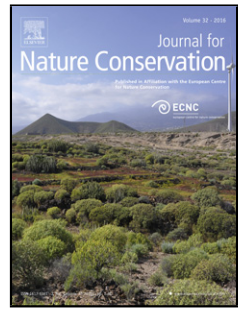
Will the Minister of ENVIRONMENT, FORESTS AND CLIMATE CHANGEbe pleased to state:-

ANSWER

MINISTER OF STATE (INDEPENDENT CHARGE) FOR ENVIRONMENT, FOREST AND CLIMATE CHANGE (SHRI PRAKASH JAVADEKAR)

(a) to (c) The Government of Gujarat requested to consider projects for Coastal Regulation Zone (CRZ) clearance in low and medium eroding coastal stretches based on rapid Environment Impact Assessment (EIA) instead of comprehensive EIA Report. The request could not be considered since a rapid EIA, which is based on one season data may not address all the environmental concerns. As per the procedure prescribed for seeking prior clearance under the CRZ Notification, 2011, all project proposals in stretches classified as low and medium eroding as well as stable coasts shall be accompanied by comprehensive EIA studies based on three season data. This requirement is uniformly applicable throughout the country. To conserve and protect such coastal stretches, promote development through sustainable manner, it is important that comprehensive EIA studies are carried out based on scientific principles and Environment Management Plans worked out accordingly before considering proposals in such stretches.

Accepted Manuscript



Title: The negative effect of dredging and dumping on shorebirds at a coastal wetland in northern Spain

Author: Juan Arizaga Juan A. Amat Manu Monge-Ganuzas

PII: S1617-1381(17)30070-5
DOI: http://dx.doi.org/doi:10.1016/j.jnc.2017.02.006
Reference: JNC 25541

To appear in:

Received date: 25-2-2016
Revised date: 8-11-2016
Accepted date: 9-2-2017

Please cite this article as: Arizaga, J., Amat, J. A., and Monge-Ganuzas, M.,The negative effect of dredging and dumping on shorebirds at a coastal wetland in northern Spain., *Journal for Nature Conservation* (2017), http://dx.doi.org/10.1016/j.jnc.2017.02.006

The negative effect of dredging and dumping on shorebirds at a coastal wetland in northern Spain.

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ABSTRACT

Dredging and/or dumping actions at coastal environments are a common phenomenon worldwide. The re-working of dumped sediments from their disposal sites to places of great ecological value can have a very strong impact on the ecosystems through deep changes over the communities and the trophic web. Using a relevant dredging-dumping episode carried out in 2003 at Urdaibai, one the chief estuary areas in northern Iberia, we tested the consequence of this action on the subsequent use of the zone by shorebirds. The surface sediment characteristics before and after the dredging and dumping actions were also compared. The dredging at Urdaibai showed a negative effect on bird abundance in three out of the eight species tested overall (dunlin, grey plover, common ringed plover). Highest-ranked models supported a decrease in their population sizes two years after the event. In this scenario, local authorities should be appealed to take dredging and dumping effects into account in order to improve the estuary management.

KEYWORDS

Aquatic bird populations; conservation biology; mudflats; population trends; sandy areas; Urdaibai Biosphere Reserve.

INTRODUCTION

All ecosystems are subject to some degree of perturbation, and all organisms are well adapted to cope with predictable perturbations, such as those determined by seasonal events. However, extreme or unpredictable perturbations, either natural (e.g. hurricanes) or owing to human activity (e.g. fires), could cause severe effects on ecosystems, from which it might take decades to recover (Borja et al. 2010; Pons and Clavero 2010; Manning et al. 2011).

The conservation of intertidal coastal environments is today a major concern for ecologists, managers, and the society in general (Weller 1999; Ma et al. 2010). Habitat loss and degradation are part of a problem that affects many intertidal wetlands all over the world (Eddleman et al. 1988; Bildstein et al. 1991). For instance, the global annual loss rate of coastal salt marshes is calculated to be 1-2% per year (Duarte et al. 2008), a rate which is above of the 0.5% per year loss rate of tropical forests (Achard et al. 2002).

Many intertidal coastal environments, mostly those linked to estuaries, have been historically used as natural harbors, an activity that is often associated with constant or periodic dredging in order to keep or increase the depth of these water bodies (Bary et al. 1997). The material (clay, sand or mud) extracted during such dredging is often dumped close to the dredging area to minimize the economical cost of the transport (Bary et al. 1997). One of the main consequences of dredging and dumping actions is habitat burial or destruction, with a negative impact on the ecosystem, especially on the macrobenthos that is situated in the bottom of the trophic network (Lindeman and Snyder 1999; Lewis et al. 2001; Boyd et al. 2005; Erftemeijer and Lewis 2006). Thus, any negative effect on such communities can alter the entire trophic structure related to the mudflats and, consequently, induce negative effects on upper trophic levels.

Clayey-muddy and sandy substrates do not host the same communities of macrobenthos that constitute the food of many shorebirds (Colwell 2010). In general, mudflats are commonly richer in shorebird food than sandy areas (Burger et al. 1997). Dredging and dumping actions carried out in estuary areas often cause habitat loss in very ecologically-sensible habitats, such as mudflats (Monge-Ganuzas et al. 2013). Thus, dumping of sand in some sensitive estuarine areas where there is an active sediment transport could cause a coverage of the mudflats and, consequently, long-lasting negative effects on benthic communities, as well as severe negative consequences for shorebirds using these areas (Piersma et al. 2001).

Here, we used retrospective analyses of dredging episodes on shorebirds' abundance and diversity in a tidal marsh, which could help to identify the consequences of dredging on shorebirds using the marsh. We predicted that relevant dredging and dumping actions may lower the capacity for shorebird populations to recover. To test this we used long-term data of shorebird censuses conducted in a site (an intertidal coastal environment located at the Urdaibai Biosphere Reserve, northern Spain) affected by a very important dredging and dumping episode. Together with this analysis, we also compared induced surface grain size trend before and after the dredging and dumping episode. We also predicted that the effect of the dredging and subsequent dumping episode should have been more severe on those species that forage mostly or only on the mudflats.

MATERIAL AND METHODS

Study area

The Urdaibai estuary is a coastal wetland located in the North of Spain. It was declared Biosphere Reserve in 1984, included within the Ramsar list in 1992, and SPA (ES0000144) and SAC (ES213007) of Natura 2000 in 2014. With ca. 945 ha, Urdaibai is used by a remarkable amount of mostly northern Euro-Siberian waterbirds (including shorebirds) that use this area either as a stopover site during migration period or as a wintering area (Galarza 1984; Garaita 2012). Shorebirds constitute a group of birds with conservation interest within the region (Galarza and Domínguez 1989; Hidalgo and Del Villar 2004). Urdaibai has suffered periodic dredging and dumping actions for the last 43 years (Monge-Ganuzas et al. 2013), with the last action occurring in 2003, when 243,000 m^3 were extracted from the main channel of the estuary and dumped in a sandy area close to the mouth. In comparison with previous dredging episodes, this last was very much larger (e.g. ca. 310% higher than the previous dredging in 1998-1999). After this dredging, wave winter storms together with tidal wave action progressively eroded the sediment and spread some sand towards upper estuary areas (Monge-Ganuzas et al. 2008) over much of the existing intertidal mudflats, the main foraging area for shorebirds within the estuary (Hidalgo and Del Villar 2004).

Data collection

In March 2003 (immediately before the dredging and dumping carried out at Urdaibai), 24 surface sediment samples were collected either by hand all along the main intertidal mudflats or from a 4 m-long vessel by a Van Veen grab (this last used to take samples along the chief estuary channel). Overall, the sampling net consisted in a 200 m each side orthogonal grid (Fig. 1). This sampling protocol was repeated in July of 2016. Samples were stored until their analysis in a laboratory (UPV/EHU).

Using a Laser diffraction particle size analyzer (Beckman Coulter counter LS 13 320), three replica of each sediment sample were analyzed (Nayar et al. 2007) and statistically integrated in order to obtain the weight percentage grain size distribution for each sample (Udden 1914; Wentworth 1922).

Census data consisted in counts (species and numbers of shorebirds) conducted during a single survey day in mid-January, coordinated by Wetlands International. Here, we considered a period spanning from 1992 to 2011. Censuses were conducted using a fixed, standard protocol, consisting in counting always from the same points, covering the same survey area and, if possible, by a same observer from year to year, during high tide. In general, due to the characteristics of Urdaibai, where birds accumulate in relatively small areas easy to survey during high tide (J. Arizaga, pers. obs.), high tide-census are recommended for counting waterbirds (but see Navedo et al. 2007).

Meteorological data (mean value for the daily mean temperatures in January) were extracted from the NOAA website (www.esrl.noaa.gov). We considered an effect of temperature because local numbers of waterbirds within the region can depend on climatic conditions at a local scale level (Navedo et al. 2007).

Data analyses

Sediment characteristics (percentage of sand and silt-clay of each sample) before and after the dredging and dumping actions at Urdaibai were compared with a *t*-test for repeated measures.

With the aim of conducting models on counts we selected those species which showed a median ≥10 individuals/year for the period spanning from 1992 to 2003 (i.e., before the dredging and dumping episode of 2003). This provided us a list of only 8 species of shorebirds to be considered within statistical models: dunlin *Calidris alpina*, purple sandpiper *C. maritima*, common ringed plover *Charadrius hiaticula*, Eurasian curlew *Numenius arquata*, grey plover *Pluvialis squatarola*, green redshank *Tringa nebularia*, common redshank *T. totanus*, Northern lapwing *Vanellus vanellus* (Fig. 2). Because of their trophic ecology these shorebirds may not depend on the mudflats in the same way, since some of them also (or mostly) forage in other habitat types (e.g. Northern lapwing, Eurasian curlew), such as the prairies and pastures surrounding Urdaibai (Navedo et al. 2013).

Moreover, we also calculated for each year the shorebird species diversity. We used for that the Shannon index (*H*'). It accounts for both abundance and evenness of all recorded species, and was calculated as: $H' = -\Sigma(p_i \times \ln p_i)$, where p_i is the proportion of species *i* relative to the total number of species (*R*, richness) (Magurran and McGill 2011).

Data were analysed using Generalized Linear Models (GLMs). Bird counts (abundance) of each species were used as object variable. We used the log-linear link function with negative binomial distribution errors for the GLMs due to the nature of the object variable (counts with over-dispersion). Additionally, we also conducted GLMs with *H*' as an object variable. In this case we used a linear link function with Gaussian errors. Overall, we considered four possible different explanatory variables: year (considered as a linear variable to test for log-linear trends in shorebird abundance), temperature (as a linear variable) and two effects that correspond to different responses of the shorebirds to the dredging episodes (for details see Table 1).

All possible models were ranked according to their small-sample size corrected Akaike (AICc) values (Burnham and Anderson 1998). Models differing in less than 2 AICc values were considered to fit to the data equally well (Burnham and Anderson 1998). In these cases, model averaging was carried out.

All analyses were run with R (R Core Team 2014), and the "lme4" (Bates et al. 2014) and "MuMIn" (Barton 2014) packages. Package "lme4" allows us to run GLMMs and "MuMIn" is used to calculate AICc values and for the model averaging procedure.

RESULTS

The percentage of sand within the estuary was observed to increase very significantly (Table 2). Along a north-south gradient, the sediment was richer in sand in the north but note the difference before and after the dredging and dumping of 2003 (Fig. 3).

The null model was the model best fitting data in seven out of the eight species tested overall (Table 3). However, in two of such species (dunlin, common ringed plover), models assuming an impact of the dredging and dumping were equally well supported. In another species (grey plover), the top model was the one assuming an effect of the dredging two years after it occurred (Table 3). Thus, overall, there were three species for which the dredging and dumping episode had an impact on their population sizes (Fig. 4). In addition, Northern lapwing population numbers and the diversity index were found to be affected by temperature (Table 3), although this effect was non-significant after model averaging (Table 4).

In those species where there was an effect of the dredging the higher-ranked model was the one where the response was observed to occur two years after the dredging; Table 3).

DISCUSSION

Dredging and dumping actions at coastal environments is a common phenomenon worldwide. The movement of sediments of different nature and its re-location in places of great ecological value can produce, however, a strong impact on the ecosystems through deep changes in the communities and the trophic nets (Sarda et al. 2000; Vanaverbeke et al. 2007). Quite often, these activities have dramatic effects on benthic communities (Powileit et al. 2006), with consequences at upper trophic levels. Using a relevant dredging episode carried out at one the chief estuary areas from northern Iberia, we observed a decrease in population size of several shorebird species which depended on mudflats to forage just one or two years after this event.

Although dredging and dumping in Iberian estuaries is common, unfortunately we have no evidence of available local information about their impact on shorebird assemblages. In a broader context, however, it is well known that dredging can have a severe negative impact on shorebirds as population size of bivalves or other potential prey is reduced, either because direct sediment extraction at foraging places (Lewis et al. 2001; Piersma et al. 2001) or because these feeding grounds are covered with sediments re-worked from dumping sites that alter invertebrate populations, as surely occurred at Urdaibai. The fact that the diversity of shorebirds remained constant at Urdaibai despite changes in abundance after the dredging and dumping episode of 2003 suggests that the most abundant species were similarly affected.

Although food availability was not analysed at our study sites our results would support the idea that the sand covering of the mudflats had a dramatic change on the macrobenthos that should be transferred to upper trophic levels (Boyd et al. 2005). Our results also show that the effect was very fast: the population size of some of the species was observed to decrease just two years after the dredging and dumping actions (with some models even also supporting an affect just a single year after the event).

Interestingly, and as predicted, Northern lapwing numbers, as well as those from other species less-dependent on marshes to forage) at Urdaibai were independent from the dredging from 2003. Northern lapwings or Eurasian curlews feed mostly in the pastures and cultivations existing around the estuary and, therefore, are little affected by dredging episodes at these wetland sites. Some shorebirds, indeed, seem to benefit from foraging in farmland habitats (Navedo et al. 2013), even if these would be subject to intensive farming practices (Lindström et al. 2010). Model selection process supported that Northern lapwings showed strong inter-annual fluctuations associated to winter temperatures at a local scale, although this effect was non-significant after model averaging, probably due to the high over-dispersion of data. The presence of this species in southern Europe is well reported to be highly stochastic (Tellería et al. 1996), and is mostly associated to dominant meteorological conditions during the winter in central Europe (SEO/BirdLife 2012). Presented results partly support the idea that the population that spends the winter in northern Iberia increases with decreasing temperatures.

The specific variable effect of temperature on bird abundances (with a positive effect in some shorebirds and a negative effect in others) along the coast of the Bay of Biscay was also reported by Navedo et al. (2007). A positive effect of temperatures on local numbers could be associated to better survival during warmer winters either due to higher food availability (Yasué et al. 2003) or to lower thermoregulation costs (Ketersen and Piersma 1987). However, local abundances of other species would be shaped by decreasing temperatures, probably associated to displacements to the coastal marshes of the Bay of Biscay from colder regions situated further north or inland (Galarza and Tellería 1985).

Resilience is the capacity of an ecosystem to tolerate perturbation without switching to an alternate state (Standish et al. 2014). Urdaibai has been subject to recurrent dredging during the last 43 years. It may be that dredged material is re-worked by the tide and wave induced currents, and this may allow the recovering of the system morphology after some years (Monge-Ganuzas et al. 2013). However, even if a system could recover after a perturbation, recurrent perturbations may lower its capacity for recovering over the long-term (Díaz-Delgado et al. 2002). Noteworthy, we observed that even in 2016, i.e. 13 years after the dredging and dumping actions carried out in 2003, the percentage of sand within the sediment have passed from a mean of 38% to 64%, with this percentage decreasing across a north-south axis (i.e., from the site where the sediment was dumped towards upper estuary areas). This result suggests that the estuary has been unable to come back to an original state before the dredging and dumping episode and it may be discussed to what extent this effect is reversible, at least short- to medium-term. The action of the waves and tide, together with the increase of the sea level (assessed to be 2 mm/year) (Leorri et al. 2013), will probably strengthen this covering of the existing mudflats by sand during next years, hence it is unlikely to expect a recovering of shorebird abundance at these areas in Urdaibai.

In this scenario, local authorities should be appealed to take the dredging and dumping effects into account in order to improve the Urdaibai estuary management because this wetland is, in fact, an important Ramsar and Natura 2000 site managed by a Governing Board composed by most regional public administrations (Basque Government, Bizkaia Council, municipalities...). Dredging activities at Urdaibai were authorized or reported by a number of public administrations, including the Basque Government (Environment Department), Bizkaia Council, Basque Water Agency and the Ministry of Environment of Spain, attending to their competences. As a part of the Urdaibai Governing Board, all such public authorities should take into consideration both the dredging and dumping effects and either promote alternative solutions or limitations to this activity if it is incompatible with the preservation of the mudflats and the occurrence of shorebirds within the area and, overall, the conservation and proper management of this wetland.

Given the sedimentary connection between the best disposal areas and the mudflats at Urdaibai probably the best decision may be to forbid both the dredging and dumping due to their dramatic consequences for the ecosystem. For instance, at Odiel estuary, in southern Iberia, dredging material is dumped in areas apart from intertidal mudflats, creating good conditions for the breeding of some species like the little tern *Sternula albifrons*, Kentish plover *Charadrius alexandrinus* and the collared pratincole *Glareola pratincola* (J. A. Amat, pers. obs.). Given the size and territory use at Urdaibai, however, these sites would be hardly available hence apparently there would be no place to dump the material extracted during dredging actions.

In conclusion, we obtained statistical data support that suggest that a strong dredging and dumping episode carried out at Urdaibai resulted in a covering of existing mudflats by sandy sediment which promoted a decrease of the population size of a number of shorebird species wintering in this area. This effect was much clearer in species more dependent on mudflats to feed, but had an apparent null impact in shorebirds that also or mainly forage in other habitat types. Thus, it is highlighted that the management of the dredging and dumping activities at Urdaibai should be improved by taking into consideration the conservation of shorebirds, among other waterbird species.

ACKNOWLEDGEMENTS

We are grateful to the people who conducted the censuses and the Basque Government that collected and provided the data. Two referees (J. Navedo and one anonymous reviewer) provided valuable comments that helped us to improve an earlier version of this work.

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Table 1. Biological meanings of the models run for each species. Abbreviations:

Models	Meaning
1. Null	Population size is constant (or fluctuates from year to year but without any particular non-random effect).
2. 2004-2011	The impact of the dredging one year after the event (i.e., from 2004 onwards) is expected to have an effect on shorebird abundance.
3. 2005-2011	The impact of the dredging two years after the event (i.e., from 2005 onwards) is expected to have an effect on shorebird abundance
4. Year	Population size co-varies log-linearly with year.
5. Temp	Population size co-varies with the mean winter (Jan.) temperature.

*We also ran four additional models by adding "temp" (additive effect) to models 2 to 4. Overall, therefore, 8 models were tested.

Table 2. Mean (±95% confidence interval) percentage of sand and mud in 25 sampling points situated all along the mudflats at Urdaibai before and after the dredging and dumping episode carried out in 2003. The percentage of gravel was zero for all samples.

Type of sediment	2003 (before)	2016 (after)	*t*-test (*P*)
Sand	38.3 ± 9.9%	64.2 ± 10.4%	4.814 (<0.001)
Mud	61.5 ± 10.1%	35.8 ± 10.4%	4.704 (<0.001)

Table 3. Ranking of the top four best-ranked models obtained for each species and the species diversity (*H'* index) in relation to their small sample size-corrected Akaike values (AICc). ΔAICc: difference in AICc values in relation to the top model. Model abbreviations as in Table 1.

Models	AICc	ΔAICc	AICc weight
Dunlin			
...null	69.1	0.0	0.38
[2005-2011]	71.0	1.9	0.14
[2004-2011]	71.2	2.1	0.14
year	71.3	2.2	0.13
Northern lapwing			
...null	59.1	0.0	0.39
...temp	60.9	1.8	0.15
[2004-2011]	61.3	2.2	0.12
[2005-2011]	61.6	2.5	0.11
Eurasian curlew			
...null	66.3	0.0	0.41
[2005-2011]	68.7	2.4	0.13
[2004-2011]	68.7	2.4	0.12
...temp	68.7	2.4	0.12
Common greenshank			
...null	61.2	0.0	0.42
...year	63.6	2.4	0.13
[2004-2011]	63.7	2.5	0.12
[2005-2011]	63.7	2.5	0.12
Grey plover			
[2005-2011]	58.8	0.0	0.27
...year	59.4	0.6	0.20
[2004-2011]	59.6	0.7	0.18
...null	60.0	1.2	0.15
Common redshank			
...null	61.2	0.0	0.42
...year	63.6	2.4	0.13
[2004-2011]	63.7	2.5	0.12
[2005-2011]	63.7	2.5	0.12
Common ringed plover			
...null	59.5	0.0	0.32
[2005-2011]	60.8	1.3	0.17
...year	60.9	1.4	0.16
[2004-2011]	61.1	1.6	0.14
Purple sandpiper			
...null	54.5	0.0	0.43
...year	57.0	2.5	0.12
...temp	57.0	2.5	0.12
... [2005-2011]	57.0	2.5	0.12
Diversity index			
null	6.2	0.0	0.41
temp	8.1	1.9	0.16
[2004-2011]	8.8	2.6	0.12
[2005-2011]	8.8	2.6	0.11

Table 4. Coefficients (*B*-parameter estimates ± SE) of best models (ΔAICc < 2) from Table 2. Abbreviations as in Table 1; (ns), non-significant coefficient. Model averaging was carried out when there were two or more models with an AICc < 2 in relation to the top model (but see comments [2] and [3]).

Species	Intercept	[2005-2011][1]	Temp
Dunlin	+0.888	-0.327	
Northern lapwing	+0.574		-0.187 (ns)
Eurasian curlew	+0.666		
Common greenshank	+0.381		
Grey plover[2]	+0.575	-1.318	
Common redshank	+0.275		
Common ringed plover[3]	+0.324	-0.706	
Purple sandpiper	-0.026		
Diversity index (*H'*)	+1.403		+0.040 (ns)

[1]Reference value ($B = 0$): period 1992-2004.
[2]Coefficients only from the top model, since the other models included alternative (but not additive) effects.
[3]Coefficients only after averaging model one and two, since the other models included alternative (but not additive) effects.

Fig. 1. Location of the sampling points considered to sample sediment characteristics all along the intertidal mudflats at Urdaibai.

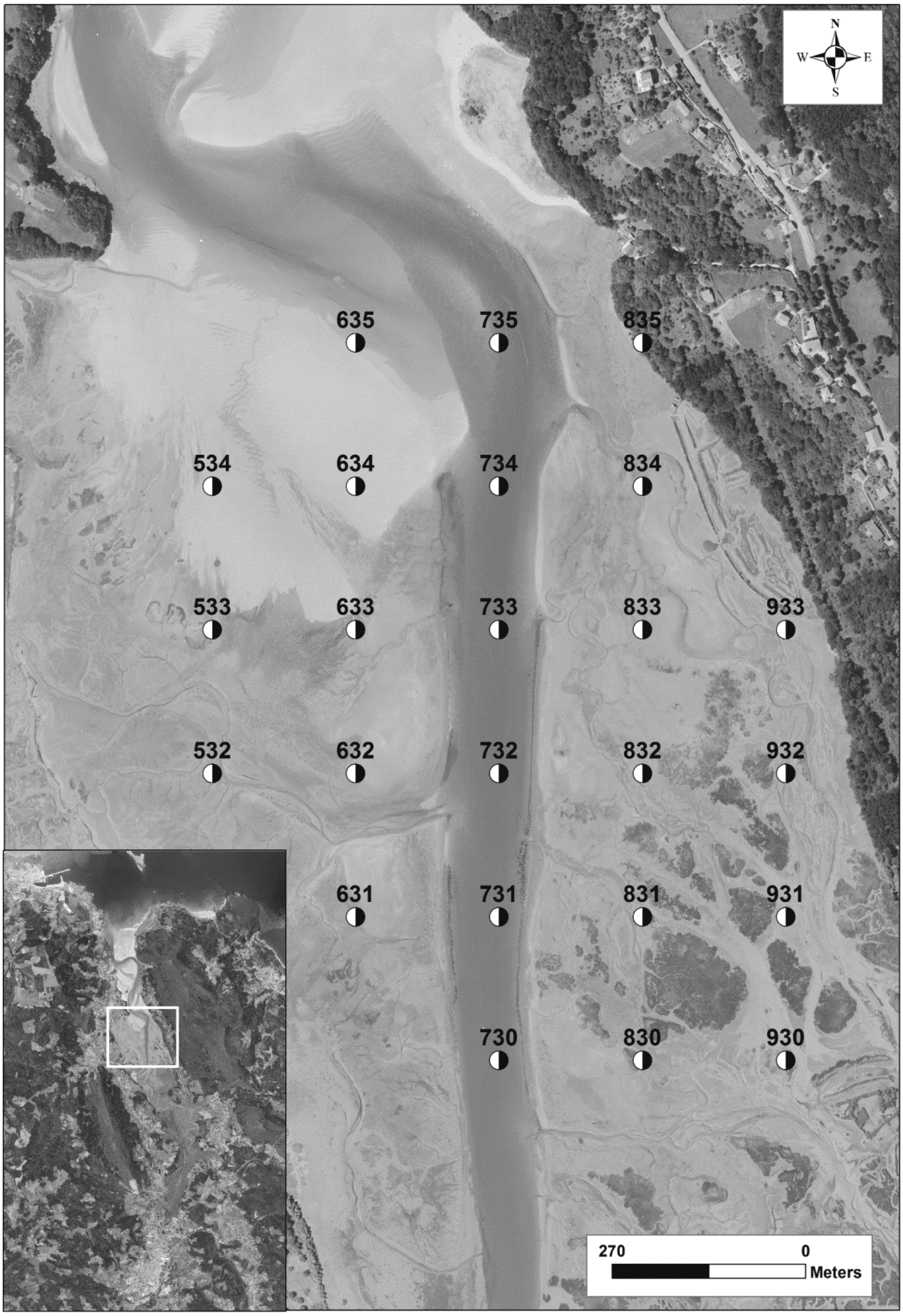


Fig. 2. Relative abundance (mean ± SD) of the ten most abundant shorebirds that overwinter at Urdaibai, period 1992-2011. Ruddy turnstones and spotted redshanks showed a median population size <10 individuals per winter for the period 1992-2003, and were not included in the analyses.

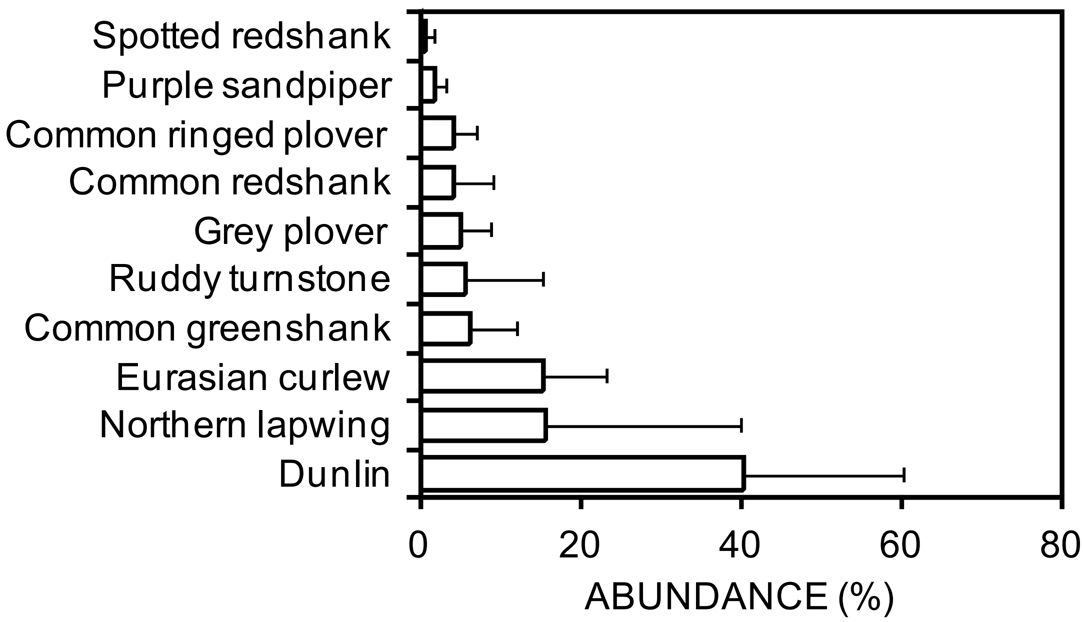


Fig. 3. Mean (±SE) percentage of sand along a north-south axis (1 stands for the sampling points 635-835 in Fig. 3; 2 for the points 534-834, etc.) of those samples taken to characterize the sediment of the intertidal mudflats at Urdaibai.

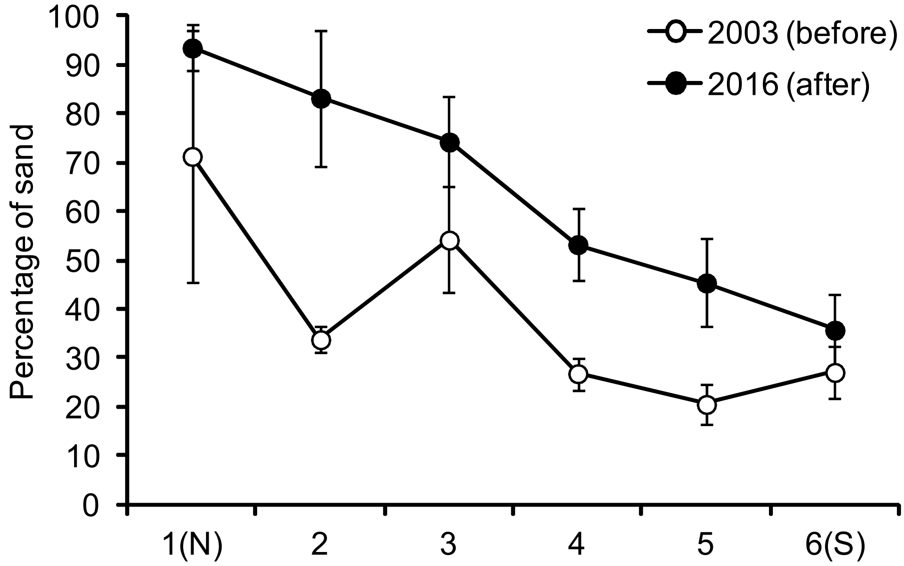
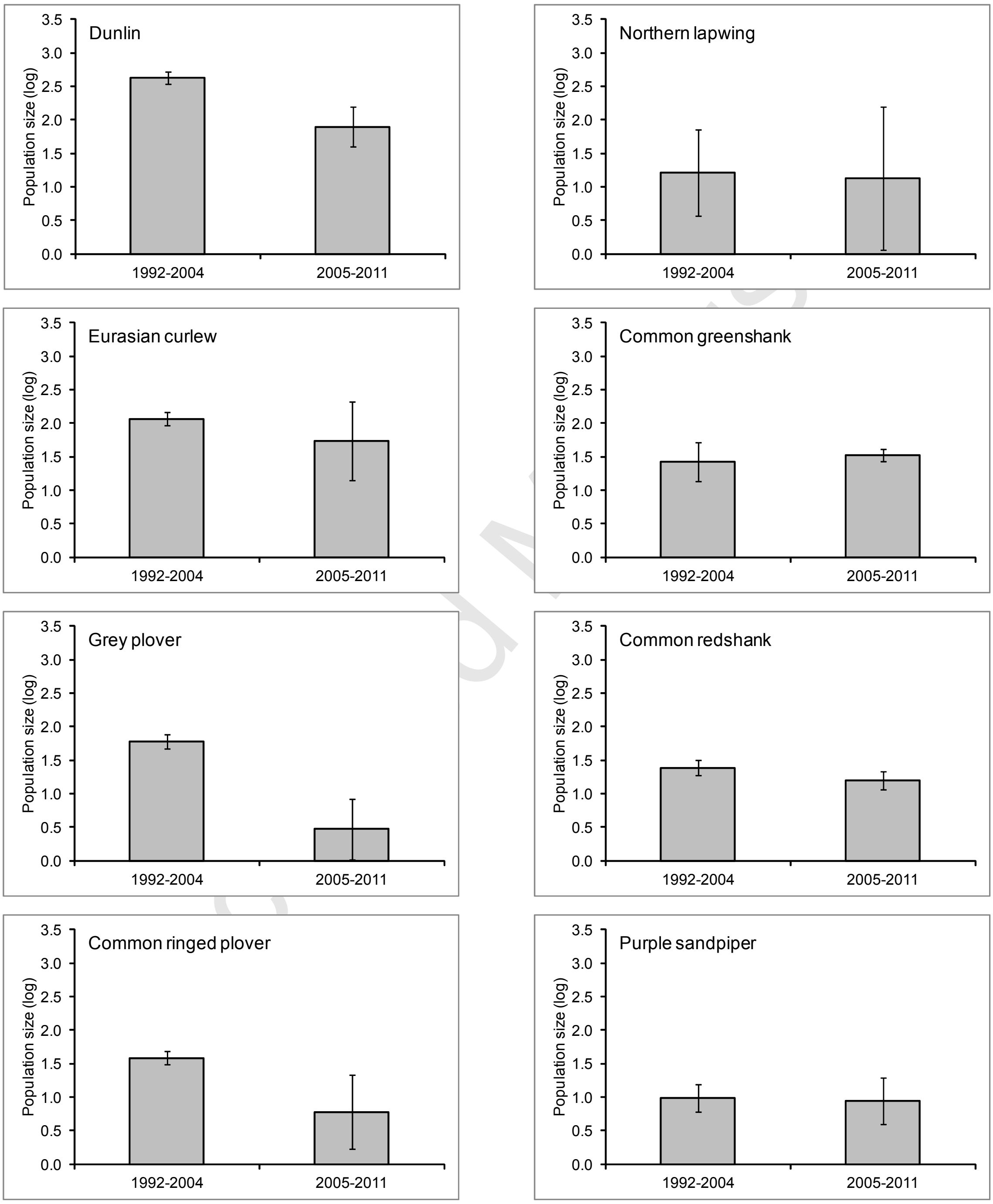


Fig. 4. Mean (±95% confidence interval) population size (log-transformed) of shorebirds before and after the dredging and dumping actions of 2003 at Urdaibai, in northern Iberia.



SETBACKS AND SURPRISES

Do restored oyster reefs benefit seagrasses? An experimental study in the Northern Gulf of Mexico

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Oyster reefs and seagrass beds are being lost worldwide at alarming rates. These habitats provide many services to humankind and, thus, much effort has been dedicated to their restoration. Here, we examine the efficacy of created oyster reefs at enhancing seagrass beds through the amelioration of hydrographic conditions and water quality. We carried out a field experiment in the Northern Gulf of Mexico where we compared areas shoreward of created reefs with adjacent reef-free areas over several years using a before-after control-impact (BACI) design. The reefs were built with oyster shell, measured 65 m, and were placed at circa 100 m from the shoreline to ensure subtidal conditions and enhance oyster recruitment. The BACI results showed few and disparate effects of the reefs, even when distance from the reef was factored in. However, we found a temporal increase in seagrass cover throughout all the experimental area (i.e. including both reef and control plots) following reef deployment. Interestingly, further analysis with satellite imagery showed the experimental area had higher seagrass cover 5 years after reef deployment than it did before reef deployment, but such increase was not observed for nearby areas. In concert, the results suggest "shadow" effects for the reefs examined, where positive effects on seagrass beds extend beyond the area directly shoreward from the reef. Oyster reef restoration may have positive impacts on shallow seagrass beds in turbid, high-energy systems; however, more work on the extent and mechanisms for this interaction is needed.

Key words: BACI design, ecosystem services, living shorelines, shoalgrass, water quality, widgeongrass

Implications for Practice

- The positive effects of reefs on seagrass beds may expand beyond the reef and "spill over" adjacent areas.
- Confirmation of "shadow effect" of restored reefs on adjacent control areas requires studies at large scales with controls areas sufficiently far from the reefs.

Introduction

The importance of shellfish reefs as nursery grounds for commercially and ecologically important fish species has been well established (Beck et al. 2001). The complex 3-dimensional reef structures provide refugia for prey species and predatory grounds for predators. Many ecosystem services are associated with shellfish reefs, namely water quality enhancement, nutrient cycling, and assimilation; benthic-pelagic coupling and protection and enhancement of adjacent emergent and submerged vegetation (Peterson & Lipcius 2003; Newell & Koch 2004; Newell et al. 2005; Plutchak et al. 2010). Despite their importance, oyster reefs are among the most exploited habitats with a global areal loss of 85% within the past 130 years (Beck et al. 2011). Within the last century, losses have amounted to 64% for reef extent and 88% for oyster biomass in the United States alone (Zu Ermgassen et al. 2012). Unsustainable harvesting practices (Rotschild et al. 1994), sedimentation (Coen & Luckenbach 2000), degraded water quality and hypoxia (Lenihan & Peterson 1998), and salinity alteration (Volety 2008) are the major causes of oyster reef degradation. Degraded reefs provide lower levels of ecosystem service than healthy reefs, which has become a major concern (Beck et al. 2011).

To mitigate the adverse impacts of oyster loss, restoration and research efforts have accelerated in the Atlantic and the Gulf coasts (Schulte et al. 2009; La Peyre et al. 2014). The Atlantic and the Gulf coast native, the eastern oyster (*Crassostrea virginica*), is one of the most commonly restored shellfish species, because it forms more extensive and complex reefs than any other bivalve species (Rotschild et al. 1994). Restored oyster reefs are considered an important "living shorelines" component, that is, an ecological alternative to traditional shoreline stabilization structures (bulkheads, riprap revetments, etc.; Scyphers et al. 2011).

Author contributions: SS, JG, RM, DB, KH, SP, CF, JC conceived and designed the research; SS, JG, RM, DB, JC performed the experiments; SS, JG, RM, DB, KH, SP, JC analyzed the data; JG, DB contributed reagents/materials/analysis tools; SS, JG, RM, DB, KH, SP, CF, JC wrote and edited the manuscript.

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doi: 10.1111/rec.12329
Supporting information at:
http://onlinelibrary.wiley.com/doi/10.1111/rec.12329/suppinfo

Success in restoration projects is often gauged as the degree to which the restored habitat provides the goods and services of their natural counterparts. Assessing restoration progress may require long-term comprehensive monitoring, which may be challenging to maintain due to budgetary and regulatory constraints (Suding 2011). Further, some restoration projects require preemptive preparatory measures (i.e. adaptive management) for unpredictable outcomes, thus complicating the progress assessment. Monitoring programs that are restricted in time and space may overlook recovered benefits from restoration programs (Bell et al. 2014). Most monitoring of restored oyster reefs has focused on enhancing shellfish and finfish abundance, along with increased market values for fisheries (Gregalis et al. 2009). Other benefits of restored oyster reefs, such as enhanced water quality and shoreline protection, have also been monitored (Plutchak et al. 2010; La Peyre et al. 2014). However, assessments of effects of restored reefs on adjacent seagrass beds are rare, although seagrasses and oysters typically co-occur in nearshore estuarine waters of Northern Gulf of Mexico (nGOM).

Seagrasses are important ecosystem engineers and provide various ecosystem services, such as, habitat provision (Heck et al. 2003); sediment stabilization (Christianen et al. 2013); wave attenuation (Fonseca & Cahalan 1992); carbon sinks (Duarte et al. 2013); and neighboring system subsidization (Heck et al. 2008). However, seagrasses are declining globally due to synergistic effects of natural and anthropogenic stressors (Orth et al. 2006; Waycott et al. 2009). Seagrass restoration typically involves seed sowing, seedling planting, and/or sod transplantation. Although, restoration techniques have improved substantially within the past decade, it remains expensive to restore seagrass beds. Further, such efforts often do not produce seagrass beds (Christiaen et al. 2013). The deployment of oyster reefs could, however, prove as an efficient technique to bring back seagrass beds into coastal systems or, alternatively, enhance existing beds shoreward of the reefs. Along with enhancing seagrass beds, oyster reefs could also generate other environmental benefits, therefore providing an "all-in-one" rather than "piece-meal" restoration approach. "All-in-one" approach considers restoration and conservation of key estuarine habitats collectively rather than focusing the conservational efforts on each habitat independently.

Oyster reefs provide physical protection on their leeward side, thereby ameliorating conditions for seagrass growth (Meyer et al. 1997; Piazza et al. 2005). Sedimentation (from buffered wave energy) and nutrient inputs (addition of feces and pseudo-feces from live oysters) may enhance seagrass growth (Newell & Koch 2004). Moreover, oysters may increase water clarity by filtering particulate organic matter, thus benefitting seagrasses (Peterson & Lipcius 2003; Plutchak et al. 2010). In this study, we evaluate the impacts of restored reefs on water quality and seagrass beds shoreward of the reefs.

We hypothesize that the deployed reefs will improve water quality (through water filtration and wave attenuation) and subsequently will enhance seagrass growth in comparison to reef-absent control areas (Fig. S1). To test these hypotheses, we deployed four oyster reefs in a coastal location of the nGOM. Our study encompassed 1.5 km of shoreline and compared the deployed reefs with adjacent control areas over more than 3 years. To assess the impacts of created reefs, we used a BACI design to compare reef and control plots. BACI designs account for ambient temporal and spatial variability between reefs and reef-free plots prior to reef deployment and include that variability into the analysis to describe more accurately the specific impacts of the reefs (Smith 2002). Such designs are useful in assessing restoration outcomes (Geraldi et al. 2009; Plutchak et al. 2010).

Methods

Study Site and Oyster Reefs

The study site was located in northeast Point-aux-Pins (NEPAP), a peninsula located in Portersville Bay in coastal Alabama (site center: 30.38°N, 88.30°W; Fig. S2A). Tides are diurnal (mean tidal amplitude: 0.43 m). Prevalent winds are from the south/southeast in spring/summer and from the north in winter accompanied by cold fronts. Mean wind speed averages 18 km/hour with high seasonal variability. Patches of seagrass, namely shoalgrass (*Halodule wrightii*) and widgeon grass (*Ruppia maritima*), occur only in shallow areas (<1 m) due to high water turbidity (Heck et al. 2001). The patches are usually small (a few m^2). Smooth cordgrass (*Spartina alterniflora*) spreads along the marsh fringes with a few intermittent escarpments. Oyster clumps are frequent through the intertidal marsh zone (Moody et al. 2013). Salinity oscillates within 20–27 ppt, although episodes of low salinity may occur.

Four reef complexes made of loose oyster shell were constructed perpendicular to the dominant wind direction in September 2009 by adding clean shells bought from a local shell-processing vendor. Our study followed a randomized paired design ($n = 4$), with each pair consisting of a reef complex and a control plot with no reef (Fig. S2B). Reefs and controls within the pairs were separated by 75 m, and adjacent pairs by 100 m (except Pairs 3 and 4: separated by 125 m). Each reef complex consisted of three trapezoidal units, each unit was 25 m long, 5 m wide, and 1 m tall (Fig. S2C). Biodegradable reinforcing fences were set up around the reefs for extra support. Reefs were constructed approximately 110 m from the shoreline (Fig. S3), where the mean lower low water depth corresponded to the height of the reefs, thereby submerging most of the reef area.

Sampling

Within each control and reef plot, three parallel transects separated by 25 m were established ($n = 24$). Each transect ran perpendicular from the shoreline to the deployed reef (or equidistantly in case of the controls). Along the three transects, fixed sampling stations were established at 1- (shore-), 55.5- (mid-), and 110-m (reef-stations) from the shoreline, for nine sampling stations within a plot (total sampling stations: $n = 72$; Fig. S3). At the nine sampling stations, water quality (total suspended solid, particulate organic matter, and water column chl-*a*), sediment (organic matter, silt-clay fraction, and benthic chl-*a*), seagrass, and polychaete samples were taken approximately monthly from April to October and once each in November, January, and March starting from December 2008. Standard sampling methods are provided as Supporting Information (Appendix S1).

Further, we obtained continuous records of water temperature, salinity and dissolved oxygen from nearby weather stations at Dauphin Island Katrina Cut (2011–2012; 16 km from NEPAP; Fig. S2A) and Dauphin Island East End (2006–2012; 25.5 km from NEPAP). Additional hydrographic data were obtained from a YSI-probe stationed at Portersville Bay (2008–2010; 5 km away YSI Model: 6600 EDS V2–4 units, YSI Inc., Yellow Springs, Ohio, USA). These hydrographic parameters were measured at 0.5 m above the sediment.

We mapped reef footprint in November 2009, October 2010, and November 2011. Shellfish densities on the reefs were measured semiannually from April 2010 to May 2012 for five times. We measured water currents using SeaHorse tilt current meters by simultaneously deploying eight current meters in all plots at the central sampling station (station 5; Fig. S3). The current meters were deployed for three time-periods spanning from March 2012 to October 2012 (24 days in March; 30 days in April/May; and 23 days in September/October).

Similarly, we measured point photosynthetically active radiation (PAR) at the central sampling station. These measurements were taken at varying intervals (weekly prereef deployment and approximately monthly from April–October and bi-monthly from November–March postreef deployment). Additionally, continuous PAR was measured at the central sampling stations from March to November 2011. Along with the seagrass measurements taken at all nine stations in each plot, we also monitored specific seagrass patches located between the reef (or adjacent control) and the shoreline for 2 years. These measurements were taken postreef deployment approximately monthly from April to October and once each in November, January, and March.

Finally, we analyzed seagrass cover along the NEPAP shores using satellite imagery (orthoimagery) acquired from Landsat (Thematic Mapper) satellite and QuickBird and Ikonos imagery obtained from Google Earth (GE). GE imagery is easily available, has high-resolution and true color composition (24-bits depth), and is frequently used to map coasts (Collin et al. 2014). Our intent was to compare seagrass cover pre- and postreef deployment between the pairs and surrounding areas south and north of the pairs to gauge evidence of "shadow" effects by the reefs. We divided the shoreline in four areas (Fig. S4), (1) a south control area (south of the experimental pairs 1 and 2, past a man-made canal); (2) pairs 1 and 2 together; (3) pairs 3 and 4 together; and (4) a north control area (north of experimental pairs 3 and 4 by mouth of Little River). The four areas share a similar physical and environmental setting.

Statistical Analysis

Reef effects were analyzed following a paired before-after control-impact (BACI) design. For each sampling time, we calculated the difference between each of the sampling stations in the reef plot and the corresponding station in the control plot in the pair (e.g. station 1 in reef plot – station 1 in control plot; station 2 in reef plot – station 2 in control plot, and so forth; Fig. S3). Then we calculated the mean value of the nine differences, and those mean values were compared before and after reef deployment with a *t* test for each pair separately. For point PAR measurements, this was done for station 5 (mid-station) because those data were only obtained at that station. To see if there was any effect of distance from the reef, we carried out similar analysis using the average values for the three closest stations to the shoreline (1, 2, and 3), the three mid-distance stations (4, 5, and 6), and the three stations closest to the reef (7, 8, and 9; "reef-proximity" specific analysis).

One-way analysis of variance (ANOVA) was used to test the difference in bivalve densities among sampling dates followed by post hoc Tukey tests. Water-current velocity and continuous PAR were only measured at station 5 during a number of time intervals postreef deployment. To analyze these data, we used two-way repeated measures ANOVA with treatment as the among-subject factor and time as the within-subject factor. For these analyses, daily averages were used. For PAR, we considered the readings between 10:00 and 16:00 to calculate daily averages. A different repeated measures ANOVA (RMANOVA) was done for each time interval. Seagrass patch measurements were also analyzed with two-way RMANOVA with treatment as the among-subject factor and time as the within-subject factor. The measurements obtained for the three patches at each plot on each sampling time were averaged into one single replicate.

The temporal dynamics of *R. maritima* and *H. wrightii* were examined using two-way RMANOVA with time as the within-subject factor and species (*R. maritima* vs. *H. wrightii*) as the among-subject factor. The additional seagrass analysis was a follow-up of the BACI results (see justification in the Results section) and specifically focused on the temporal trends of both species in the study area. We pooled reef and control plots and considered each of the nine sampling stations within each plot as a single replicate. Post hoc paired *t* tests comparing density between the two species were done between control and reef plots separately for each sampling time when there was a significant interaction between time and treatment. We did these analyses for each pair separately.

Prior to ANOVA analysis, data were tested for normality using the Shapiro–Wilk test and homogeneity of variance using the Bartlett's test, and transformed when necessary to meet the assumptions of the test. We also confirmed that the sphericity requirement for RMANOVA was met. All statistical tests were considered significant at $p \leq 0.05$. All statistical analyses were done using SigmaPlot version 12.0.

Results

Overall, we recorded similar mean values and ranges of water temperature, salinity, and dissolved oxygen in the three nearest

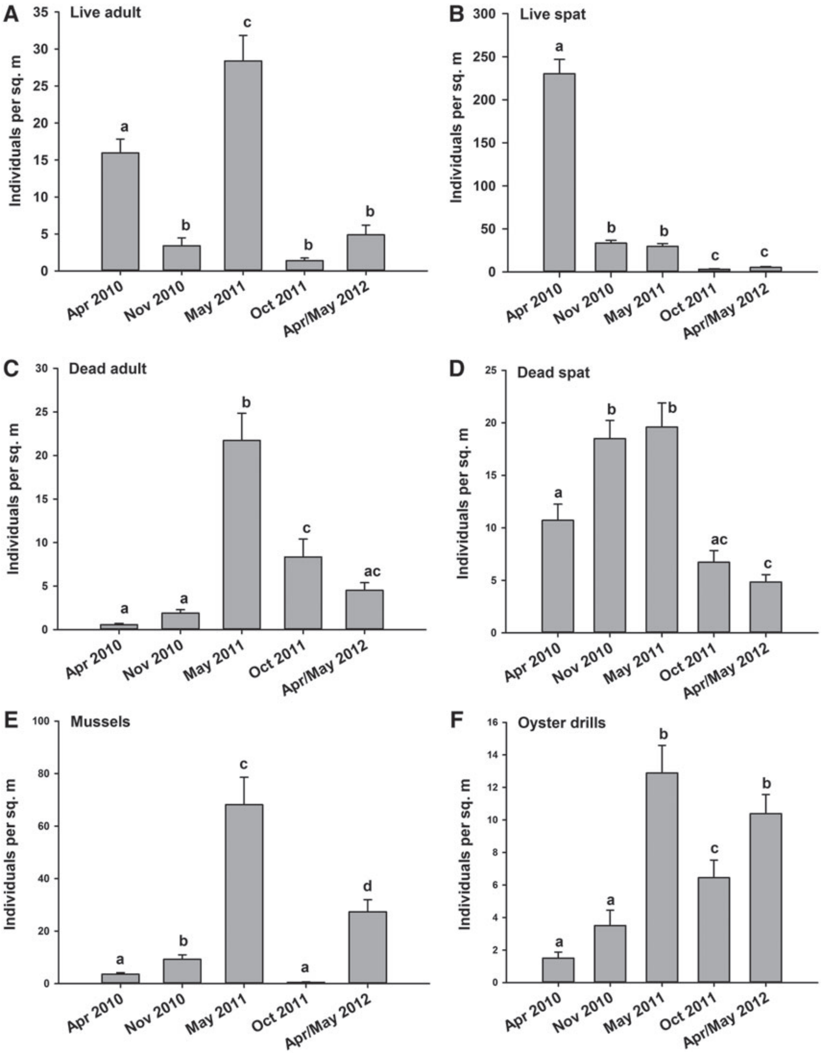

Figure 1. Density of (A) live adult oysters (SH > 3 cm); (B) live spat (SH < 3 cm); (C) dead adult oysters; (D) dead spat; (E) mussels; and (F) oyster drills. Letters denote significant differences following Post hoc Tukey tests. Bins represent means and bars ± SE.

hydrographic stations to NEPAP (Table S1). Water temperature values showed typical seasonal oscillations. Salinity remained most often within typical values for open coastal waters in the area, although large decreases to oligohaline levels occurred throughout the study period. Generally, waters remained well oxygenated, but hypoxic/anoxic levels were recorded in the stations. Our study site was very shallow and generally well mixed and, thus, anoxic/hypoxic conditions in the water column should have occurred rarely.

Densities of live and dead adult oysters were highly variable, with the peak in May 2011 (Fig. 1A & 1C). Lower adult oyster densities later in the sampling periods could be due in part to the decline in recruitment. The live spat density decreased consistently through the sampling period to <20 individuals m^{-2} on the last two sampling dates (Fig. 1B). The dead spat density was also low on the last two sampling dates (Fig. 1D). High oyster-drill densities were found later in the sampling period (Fig. 1F). Mussel densities also oscillated through the sampling period (Fig. 1E). Our reef footprint maps showed that the reefs were stable and stayed in place (Fig. S5).

We observed optimal water temperature, salinity, and dissolved oxygen; however, water-current velocity, total suspended solid (TSS), particulate organic matter (POM), chl-*a*, and PAR values did not show significant difference between reef and control plots (Tables S2, S4, S6; Figs. S6, S7A, S8). This was also the case for the separate "reef-proximity analyses," that is stations pooled as 1–3, 4–6, and 7–9 (Table S5; Fig. S3). Water depths at station 5 were similar among plots (Table S7).

Similarly, we did not observe significant impact of reef deployment in sediment metrics. (Table S4, S5; Fig. S7B). However, when all nine stations were considered, we found a significant impact of reef deployment in Pair 3, with higher polychaete density observed in the reef than in the control plot after reef deployment but not before reef deployment (Table S4; Fig. S7B). These across-plot differences for polychaete density

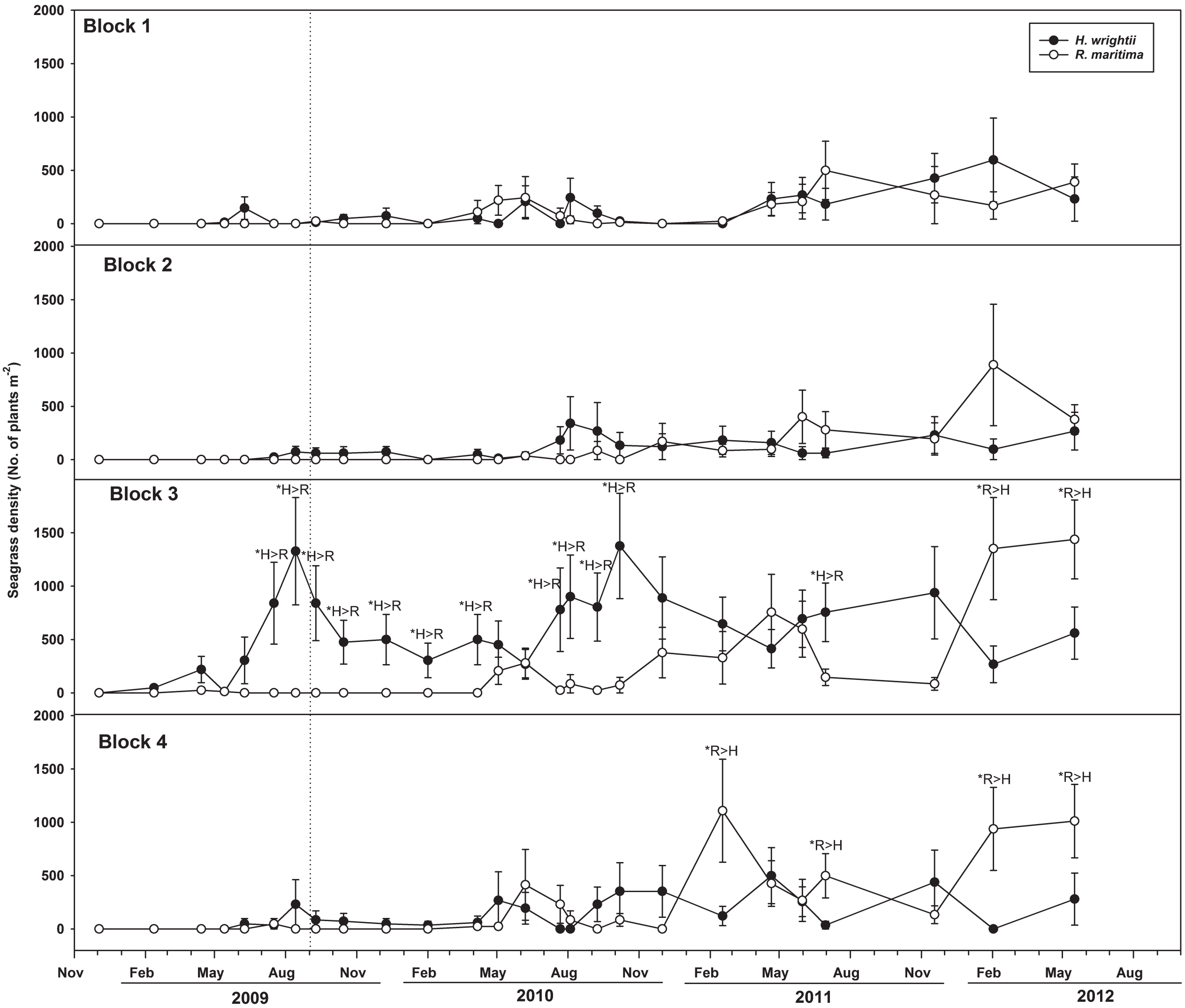

Figure 2. Seagrass density temporal dynamics. Bars represent ± SE. Dotted line represents the time of reef deployment.

were probably driven by the three stations closest to the reef (Table S5).

When all nine sampling stations were considered in seagrass analysis, few and disparate significant impacts of reef deployment were found for density and biomass after reef deployment (Table S4, Fig. S7C). Reef-proximity analyses also showed similar effects (Table S5). Density within the patches changed over time (RMANOVA; $p < 0.05$ for time effect) but not between control and reef plots ($p > 0.05$ for treatment and interaction effects) for both seagrass species (Fig. S9).

Despite finding no significant impact of reef deployment on seagrass abundance using the BACI analysis, we noticed an overall increase in seagrass abundance after reef deployment across the area examined. Seagrass density generally increased over time after reef deployment in all four pairs (Table S8; Fig. 2). In Pairs 1 and 2, the pattern of increase was similar for *Halodule wrightii* and *Ruppia maritima,* and density did not differ between the two species during the experiment. In Pair 3, *H. wrightii* increased before reef deployment and showed higher densities than *R. maritima* during most of the study period. *Ruppia maritima* increased later and had higher densities than *H. wrightii* on the last two sampling dates. In Pair 4, the increase after reef deployment was similar for the two species in the first half of the experiment, but *R. maritima* increased to a larger extent and was often denser than *H. wrightii* during the second half of the experiment.

Our remote sensing analysis revealed that, 5 years after reef deployment, seagrass cover was higher than prereef deployment levels throughout the entire area covered by the experimental pairs (including both the control and reef plots). Seagrass cover area at Pairs 1 and 2 increased by 5.4 times from 2,613 m^2 (in June 2006) to 13,998 m^2 (in November 2013) and at Pairs 3 and 4 it increased by 14.1 times from 3,589 m^2 (June 2006) to 50,662 m^2 (November 2013). However, seagrass cover area at South Control decreased by 0.6 times from 3,363 m^2 (in June 2006) to 2,057 m^2 (in November 2013) and at North Control it

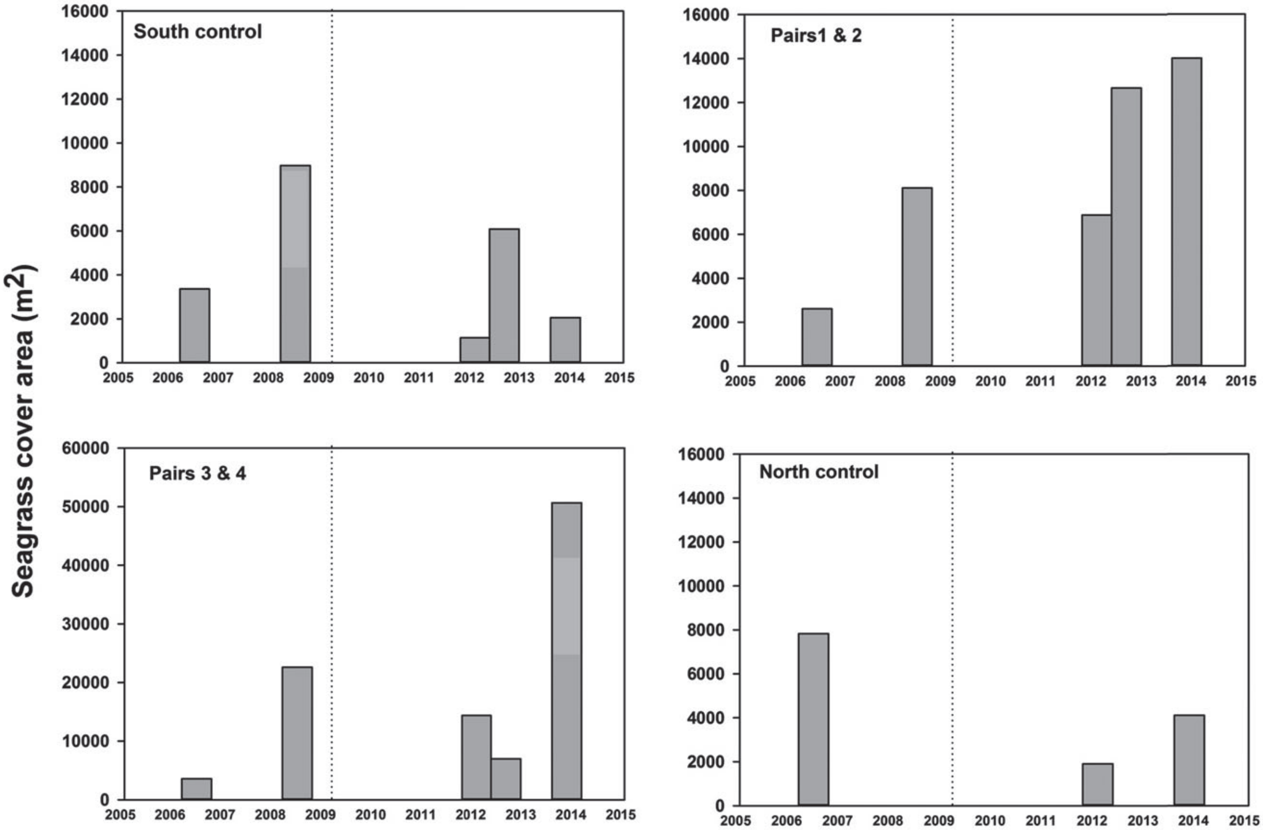

Figure 3. Seagrass cover in our experimental and adjacent areas. Dotted line represents the time of reef deployment.

decreased by 0.5 times from 7,823 m^2 (June 2006) to 4,100 m^2 (November 2013) (Fig. 3).

Discussion

Although oyster reefs are known to provide habitat for fish and crustaceans, yet their role in facilitating adjacent habitat expansion is less explored. Here, we report that created oyster reefs could potentially enhance seagrass cover in shallow waters in the nGOM.

Stability of restored reefs is crucial for quality ecosystem services and sustained oyster population, which may be dependent in part on availability of substrate for new spat recruitment (La Peyre et al. 2014). Hydrographic records generally showed favorable conditions for oyster growth around NEPAP and the created reefs maintained the structural integrity. Despite the prediction of high spat recruitment from physical and biological models around NEPAP, favorable hydrographic conditions and stable reefs did not translate into high oyster density (Hoese et al. 1972; Kim et al. 2010). Indeed, mean live oyster density of 35 individuals m^{-2} in this study represents only a fraction of values reported elsewhere, e.g. 728 ± 101.9 in Louisiana (La Peyre et al. 2014), 200–300 in Alabama (Geraldi et al. 2009), and 250.4–1026.7 in Chesapeake Bay (Schulte et al. 2009).

We suggest that predators played a role at our experimental site. Adapted to high salinity, oyster drills are one of the natural predators of the eastern oyster in the nGoM (Garton & Stickle 1980). Indeed, oyster drill density was highest in 2011, followed by 2012, corresponding with the most saline years around NEPAP. Other mobile predators of the eastern oysters such as black drums and blue crabs are not uncommon in the Gulf waters, and NEPAP is no exception (Brown et al. 2008; Gregalis et al. 2008). Decreased spat recruitment in the later sampling events coinciding with high mortality may be due to predation and/or physical disturbance (Scyphers et al. 2011). Predation, mortality, and physical disturbance probably resulted in low oyster density at our reefs.

Based on established premises, we expected that reefs abundant with live oysters would attenuate wave energy and increase water clarity, and consequently have positive impacts on the light-limited seagrass beds leeward of the reefs (Newell & Koch 2004). However, the BACI analysis did not support large or consistent reef effects on water quality and seagrass beds. This was also the case when distance from the reef was factored in and reef-proximity was analyzed. Based on these results, our reefs did not perform as expected. Limited wave attenuation efficiency of subtidal reefs coupled with low bio-filtration from sparse oysters resulted in nonsignificant reef effect on water-current velocity and water quality. Although reefs were deployed to coincide with the mean lower low water (to maximize larval recruitment), the intended recruitment level was not attained as predicted by previous studies (Hoese et al.

1972). The ratio of reef length to the reef deployment distance was probably too low to attenuate wave energy substantially. In wave-dominated open systems, such as NEPAP, frequent resuspension and constant intermixing of water may demand higher bio-filtration efficiency from large oysters (La Peyre et al. 2014); because our reefs were not densely populated, bio-filtration was limited. Further, an effect of live oysters on seagrass patches as modeled by Newell & Koch (2004) was not detected, perhaps because feces/pseudo-feces did not reach the seagrass sediment or accumulated away from the segrass beds (Booth & Heck 2009).

Seagrass patches restricted to the shallow waters of NEPAP are typically subjected to environmental stresses related to water temperature, wind speed, turbidity, and water-current velocity. Further, the shallow seagrass patches are often exposed at low tides during winter, imposing additional stress. We hypothesized that reefs would ameliorate the stressful conditions and have a positive impact on seagrasses by improving hydrological and water quality conditions. Despite the lack of support for this hypothesis, our study offers evidence that the constructed reefs may benefit the seagrass beds between the reefs and shoreline. Namely, we generally found an increase in seagrass abundance after reef deployment throughout the experimental area including all pairs and reef and control plots. These results suggest a "shadow" effect, that is, their effect may have cascaded to both sides of the reef toward the shore, thereby spilling over into the adjacent control plots.

The possibility of a "shadow" effect is supported by seagrass cover from satellite imagery in all four control and reef plots and two nearby areas on a number of dates before and after reef deployment. The nearby areas were also located in NEPAP and had similar environmental conditions to our experimental area, but were sufficiently far from the reefs to prevent "shadow" effects, at least to the same magnitude as they could have occurred in control plots adjacent to reef plots. Our control and reef plots had higher seagrass area 5 years after reef deployment than they did before reef deployment; however, such increases were not observed in nearby areas. These long-term, large-scale results also suggest that accurate assessment of oyster reefs impacts on environmental quality and status of surrounding habitats may require monitoring that extends well beyond the reefs and over several years (Bell et al. 2014). Thus, we cannot conclude unequivocally that the increase in seagrass cover after reef deployment is due to the reefs. We conclude that oyster reef restoration may have positive impacts on shallow seagrass beds in turbid, high-energy systems, but more work on the extent and mechanisms is needed. However, we iterate that significant effects of reefs could be found due to small improvements in water quality and shear (friction, wave energy), given the high natural levels of murkiness and wave energy that exist in the area (i.e. stringent limiting light and wave energy conditions for seagrass growth). To be able to detect this effect, we possibly need to work at large spatial scales with control areas sufficiently far so they are not "shadowed" by the reefs.

Acknowledgments

We express our thanks to L. Biermann, J. Gulbranson, J. Hemphill, J. Howard, J. Kay, S. Kerner, A. Macy, L. Marino, M. Metcalf, J. Reynolds, A. Schat, L. Schumacher, and K. Watson for their help in the field and the lab as well. We are extremely thankful to V. Sheremet, WHOI for his assistance in SeaHorse water-current calculations. Thanks are also due to V. Amaral and J. Zedler and two anonymous reviewers for their suggestions that greatly improved readability of this manuscript. This project was funded by ADCNR, State Lands Division (Grant # SLD-EDRP-DISL-08-1).

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Supporting Information

The following information may be found in the online version of this article:

Table S1. Hydrographic parameters (mean ± SE) at nearby stations (see text for details). Mean values correspond to the average of all daily means recorded during the period. Values in the parentheses represent the range (min–max values)
Table S2. Two-way repeated measures ANOVA for water-current velocity (cm/second).
Table S3. Before-after, control-reef values of water quality, K_d, sediment metrics, polychaete density, and seagrass metrics for (A) Pairs 1 & 2 and (B) Pairs 3 & 4. Values represent the mean ± SE for the control or reef plot before or after reef deployment. Values in parentheses represent the range.
Table S4. BACI results for water quality, K_d, sediment metrics, polychaete density, and seagrass metrics including all plot stations (see text for details).
Table S5. BACI results for water quality, K_d, sediment metrics, polychaete density, and seagrass metrics including only the three closest stations to shore (“Shore,” stations 1–3); mid-distance stations (“Mid,” stations 4–6); and farthest stations from shore (“Reef,” stations 7–9) (see text for details); ns, nonsignificant; +, positive reef effect; −, negative reef effect.
Table S6. Two-way repeated measures ANOVA for underwater bottom irradiance (μmoles/m^2/second).
Table S7. Water depth at the location (station 5) where bottom PAR measurements were recorded. Mean and range values have been calculated from depth measurements taken along with PAR point measurements at the station (see text for details).
Table S8. Two-way repeated measures ANOVA for seagrass density temporal dynamics.
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Coordinating Editor: Joy Zedler

Received: 6 July, 2015; First decision: 5 September, 2015; Revised: 2 December, 2015; Accepted: 2 December, 2015; First published online: 25 January, 2016

Ecological Functions of Shallow, Unvegetated Estuarine Habitats and Potential Dredging Impacts (with emphasis on Chesapeake Bay)

by Gary L. Ray

PURPOSE: The U.S. Army Corps of Engineers (USACE) is faced with increasing numbers of requests to dredge shallow, unvegetated estuarine waters (Figure 1). Most of these proposals involve tidal waters ranging in depth from mean low water (MLW) to 1.2 m (4 ft) below MLW, with projected post-dredging depths of 1.8 m (6 ft) below MLW. This technical note summarizes what is known about the ecological functions of these habitats and the potential impacts of dredging.

Figure 1. Aerial view of an estuary (Photo source: USACE)

BACKGROUND: In 1992, concern over potential deleterious impacts of dredging on shallow, unvegetated estuarine habitats led the U.S. Army Engineer District, Norfolk (CENAO) to request the U.S. Army Engineer Research and Development Center (ERDC) to review the ecological definition, functions, and potential sensitivity of these habitats to dredging. At that time, there was no clear definition of the term "shallow-water" and few studies specifically addressing differences in ecological functions along the depth range of interest. Since this time, there has been a concerted effort to define "shallow water" (e.g., Reilly et al. 1996) and a number of conferences have explicitly addressed ecological functions and importance of shallow-water habitats.

Definition of Shallow Water. In 1993 the U.S. Environmental Protection Agency surveyed environmental managers, regulators, and researchers in order to arrive at a consensus of what constitutes shallow water (Spagnolo et al. 1994, Reilly et al. 1996). Opinions varied widely among the participants, the agencies and geographic regions they represented, and the disciplines in which they worked. This was not surprising since "shallow" is a relative term, differing with the context in which it is used. For example, what is considered shallow by an oceanographer working on the continental shelf is quite different from someone working in estuaries or bays. After considerable debate, shallow water was operationally defined as less than 4 m (13 ft) below MLW. This definition was not meant to be an inflexible standard, but a starting point for further discussions and refinement. Despite an exhaustive search of the technical and scientific literature, no alternative definitions have been put forth since these publications.

Report Documentation Page	*Form Approved OMB No. 0704-0188*

1. REPORT DATE	2. REPORT TYPE	3. DATES COVERED
DEC 2005	**N/A**	**-**

4. TITLE AND SUBTITLE	5a. CONTRACT NUMBER
Ecological Functions of Shallow, Unvegetated Esturaine Habitats and Potential Dredging Impacts (With Emphasis on Chesapeake Bay)	5b. GRANT NUMBER
	5c. PROGRAM ELEMENT NUMBER
6. AUTHOR(S)	5d. PROJECT NUMBER
	5e. TASK NUMBER
	5f. WORK UNIT NUMBER
7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES) **U.S. Army Engineer Research and Development Center (ERDC) Construction Engineering Research Laboratory (CERL) PO Box 9005 Champaign, IL 61826-9005**	8. PERFORMING ORGANIZATION REPORT NUMBER
9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES)	10. SPONSOR/MONITOR'S ACRONYM(S)
	11. SPONSOR/MONITOR'S REPORT NUMBER(S)

12. DISTRIBUTION/AVAILABILITY STATEMENT
Approved for public release, distribution unlimited

13. SUPPLEMENTARY NOTES
The original document contains color images.

14. ABSTRACT

15. SUBJECT TERMS

16. SECURITY CLASSIFICATION OF:			17. LIMITATION OF ABSTRACT	18. NUMBER OF PAGES	19a. NAME OF RESPONSIBLE PERSON
a. REPORT	b. ABSTRACT	c. THIS PAGE			
unclassified	**unclassified**	**unclassified**	**SAR**	**13**	

Standard Form 298 (Rev. 8-98)
Prescribed by ANSI Std Z39-18

Estuarine Habitat Classifications. There are four major habitat classification schemes encompassing estuarine habitats of the United States: Shaw and Fredine (1956), Cowardin et al. (1979), Dethier (1992), and Allee et al. (2000). Shaw and Fredine (1956) is a descriptive scheme that differentiates among regularly and irregularly flooded intertidal habitats and open-water (subtidal) habitats, but provides no further distinction based on depth. Cowardin et al. (1979), Dethier (1992), and Allee et al. (2000) are all framed as dichotomous keys and share a similar hierarchical structure. The earliest of these, Cowardin et al. (1979), was developed as a tool to classify emergent wetlands. It has three general levels (and sublevels): system, class, and modifiers. As in Shaw and Fredine (1956), shallow-water habitats are distinguished under the subsystem level as either intertidal or subtidal, but there is no further distinction or definition based on depth. Dethier (1990, 1992) modified the Cowardin scheme for use in Washington State. She added an additional level, energy/exposure, to the hierarchy, permitting description of habitats as exposed, partially exposed, semi-protected, and protected. Marine habitats are additionally classed as high, moderate or low energy and estuarine habitats as open, partially enclosed, or channel/slough. Depth is incorporated as a habitat modifier for intertidal systems as backshore (areas above mean spring high water or MHWS) and eulittoral (areas below MHWS). Subtidal systems are described as shallow (extreme low spring water, ELWS, to less than 15 m) and deep (more than 15 m below ELWS). Allee et al. (2000) have further refined Cowardin et al. (1979) by restructuring the scheme and adding geographic, regional, geomorphic, topographic, and hydrodynamic levels. Of specific interest is the definition of offshore habitats by depth. Shallow offshore waters are defined as less than 200 m. Unfortunately, no distinction is made between habitats of inshore waters.

Jay et al. (2000) reviewed classification of estuaries and their habitats and concluded that a geomorphic approach would be more appropriate than the descriptive approach common to existing classifications. A hydrogeomorphic approach has previously been successfully employed for classification of wetlands (Brinson 1993). Whether or not habitats in the depth range of interest would be more effectively classified by such an approach is unclear at this time.

Shallow-Water Habitats. Shallow, unvegetated, estuarine habitats are presently defined primarily by their relative position within the gradient of tidal exposure (intertidal or subtidal), sediment type (e.g., mud or sand), and presence of biogenic structure (e.g., oyster shell). The depth range of interest is primarily subtidal, although it does represent a gradient of intermittent exposure at higher intertidal elevations. Beyond the terms intertidal and subtidal, there is no official designation of habitats by elevation (depth). Definition of habitats by sediment type also represents a gradient of conditions ranging from unconsolidated mud to gravel. In general, habitats are defined by dominant sediment type, i.e., sand, mud, muddy sand, etc. In Chesapeake Bay, nonvegetative biogenic structure is limited to oyster shell, whereas in other systems mussel and clam beds, worm reefs, and coral reefs may be present. Habitats can also be classified by their salinity: oligohaline, 0.5-5; mesohaline, 5-18; polyhaline 18-30; euhaline, above 30. Therefore, the major shallow, unvegetated, habitats encountered in Chesapeake Bay include oligohaline, mesohaline, polyhaline, or euhaline mud and sand flats, subtidal mud or sand bottoms of both tidal creeks and open bay waters, and oyster reefs.

Ecological Functions

Tidal flats, tidal creeks, and shallow subtidal bottoms. Appreciation of the ecological functions of shallow-water habitats has increased significantly during the past ten years. The ecological functions of tidal flats and shallow-water habitats include high primary production by benthic microalgae (e.g., diatoms), nutrient regeneration, decomposition of organic matter, secondary production by infauna (benthic invertebrates), feeding habitat and predation refuges for post-larval fishes and invertebrates, and feeding habitat for shore birds and wading birds (Peterson and Peterson 1979; Diaz et al. 1982a, 1982b; Diaz and Schaffner 1990; Short et al. 2000) (Figure 2).

Figure 2. The Blue crab (*Callinectes sapidus*), an important species of shallow water (Photo courtesy of Southeast Regional Taxonomic Center)

MacIntyre et al. (1996) reviewed the scientific literature on primary production of shallow-water benthic algae, finding that benthic microalgal biomass and production often equal that of the overlying water column, indicating that it plays an important role in system productivity. Pinckney and Zinmark (1993) have shown that microalgae of short-form *Spartina alterniflora* marsh and mudflats supply 45 percent and 22 percent, respectively, of total primary production at North Inlet, South Carolina. Shallow-subtidal (defined as less than 1 m) bottoms provide an additional 13 percent of production. Sand flats produce less (3 percent of total) than mudflats, as has also been reported by Barranguet et al. (1998). In addition to their contribution to primary production, microalgae are also important to nutrient cycling (Tyler et al. 2003) and to higher trophic levels (see review by Miller et al. (1996)). Middelberg et al. (2000) have shown that carbon derived from microalgae was incorporated into bacteria, meiofauna, and macrofauna in proportion to algal biomass, emphasizing its importance to estuarine food webs.

Benthic microalgae also are important in maintaining sediment stability. Underwood and Patterson (1993) demonstrated this attribute by purposefully eradicating diatoms from a patch of mud flat, thereby substantially increasing erosion in the treated patch. Similar conclusions have been reached by Austen et al. (1999) and Madsen et al. (1993) for intertidal and subtidal sediments, respectively. Meadows et al. (1994) reported that fungi provide a similar function in sandy intertidal sediments.

Infaunal communities of southeastern tidal creeks have been described by Lerberg et al. (2000) and Holland et al. (2004). Depths in these waterways ranged from a few centimeters to 1.5 m. Only three benthic species displayed distinct patterns of abundance within the creeks. Densities of the oligochaete *Monopylephorus rubroniverus* and the polychaete *Capitella capitata* were highest in the upper reaches of the creeks, while abundance of the polychaete *Heteromastus filiformis* was highest in the lower reaches. Infaunal community structure and habitat quality were degraded in watersheds with substantial commercial and urban development due to chemical pollution associated with runoff. Gillet et al. (2005) estimated secondary production of *M. rubroniverus* in two of the same waterways sampled by Lerberg et al. (2000) and Holland et al. (2004) and report that production was greater in upper than in lower reaches. Benthic algal production (as estimated from chlorophyll A concentrations) was also higher in upper than lower reaches.

Ewing and Dauer (1982) monitored benthic macroinvertebrate populations of shallow waters in the lower Chesapeake Bay and found that shoal and tidal creek stations tended to have fewer species, but three to five times more animals than deeper sites. West (1985) monitored tidal creek infauna of the Pamilco River estuary, North Carolina along a depth gradient ranging from 0.3 m to 1.5 m (1 to 5 ft) and also found that shallowest stations had the highest densities of infauna. Tourtellotte and Dauer (1983) reported that differences in benthic assemblages of Lynhaven Bay and surrounding areas sampled at depths between 2 m to 4.9 m (6 ft to 16 ft) were related principally to sediment type rather than depth. Chester et al. (1983) examined Newport River estuarine sites with depths of approximately 1 m and found that sediment type and salinity were the dominant factors determining abundances.

Shallow-water habitats are important nursery areas for post-larval and juvenile fish and shellfish. In Chesapeake Bay, young-of-the-year spot, silver perch, sea trout, and croaker dominate the nekton of waters 3.7 m or less (less than 12 ft) in spring and summer (Chao and Musick 1977). Weinstein and Brooks (1983) also found high densities of juvenile spot in tidal creeks in Chesapeake Bay. Flounder and blue crabs were also collected throughout shallow waters in that study. Both summer and southern flounder post-larvae have been shown to concentrate on Chesapeake tidal flats (Burke et al. 1991). In Cape Fear, North Carolina, tidal creeks and the fringes of marshes have high abundances of postlarval and juvenile spot, striped mullet, menhaden, and brown shrimp (Weinstein 1979). Allen and Barker (1990) describe subtidal creek bottoms at similar depths in South Carolina as important nursery habitats for larval spot, gobies, bay anchovy, and croaker. Burton et al. (1985) specifically recommend sampling for freshwater and oligohaline ichthyoplankton in shallow waters of Upper Chesapeake Bay. Hodson et al. (1981) and O'Neil and Weinstein (1987) have also documented the utilization of North Carolina and Chesapeake tidal creek habitats as feeding habitat for juvenile spot. Ryer (1987) and Orth and Montfrons (1987) examined utilization of tidal creeks and other shallow-water habitats in Chesapeake Bay as feeding habitat for blue crabs. Pihl et al. (1991) have also noted that fish and invertebrates migrate from the channel habitats into shallower, more oxygen-rich, habitats during hypoxic events in Chesapeake Bay. In contrast to widely held assumption that unvegetated habitats represent poorer nursery habitats than seagrass, Seitz et al. (2005) has recently shown that growth of juvenile blue crabs was greater in unvegetated mud and sand flats of the upper York River than the same habitats or seagrass beds in the lower river. In a companion study Lipcius et al. (2005) report that survival and overall abundance of juvenile blue crabs were also greatest in the unvegetated habitats of the upper river.

Clear evidence in support of the importance of shallow waters as a refuge from predation for both juveniles and small species of fish and decapods in Chesapeake Bay was provided by Ruiz et al. (1993). Sampling three depth zones, 1-35 cm, 36-70 cm, and 71-90 cm, they found that small species such as grass shrimp (*Palaemonetes pugio*), sand shrimp (*Crangon septemspinosa*), mummichog (*Fundulus heteroclitus*), striped killifish (*F. majalis*), the mud crab *Rithropanopeus harrisi*, four-spine stickleback (*Apeltes quadricus*), and naked goby (*Gobiosoma bosci*) were most abundant at depths less than 70 cm (2.2 ft). With the exception of the sand shrimp, small individuals (juveniles) of each species were also more abundant at depths less than 70 cm. Large species, especially predators such as blue crab (*Callinectes sapidus*), spot (*Leiostomus xanthurus*), and Atlantic croaker (*Micropogonias undulatus*) were generally most abundant at depths greater than 70 cm. Experiments with tethered shrimp, fish, and crabs showed that survival decreased with increased depth, suggesting higher rates of predation in deeper water. Additional support for the predation refuge hypothesis comes from the work of Paterson and Whitfield (2000). Sampling

South African salt marsh tidal creeks, they found that total fish abundance increased with depth in habitats ranging from less than 1 m to approximately 3 m. However, abundances of large predatory fish were disproportionately lower in depths less than 1m.

Shallow water also serves as an important feeding habitat for birds. Shorebirds and large wading birds obviously utilize the shallowest portions of the depths of interest while skimmers, kingfishers, terns, and gulls will feed in the deeper areas (Diaz et al. 1982b). The best feeding conditions will presumably be in the shallowest portions where prey will be more visible and concentrated. Scotts (1985) specifically identified mud flats, subtidal depths to 2 m (6 ft), and 2-m to 14-m depths as the most important for Chesapeake Bay waterfowl. In a review of the importance of shallow waters to waterbirds and shorebirds along the Mid-Atlantic coast, Erwin (1996) pointed out that intertidal flats are particularly important to migrant shorebirds and waters less than 2 m in depth are important to waterbirds. Fewer waterbirds are found in depths greater than 2 m.

Figure 3. Intertidal oyster bar (Photo source: USACE)

Oyster reefs. Considerable advances have recently been made in understanding the ecological functions of oyster reef habitat (Coen et al. 1999, Coen and Luckenbach 2000) (Figure 3). Long valued as a commercial resource, large masses of oysters exert a significant influence on water quality, phytoplankton productivity, and nutrient cycling of estuaries (Dame 1996). For instance, Dame and Libes (1993) found decreased concentrations of ammonium, total nitrogen, total dissolved phosphorus, and total phosphorus in tidal creeks where oysters had been removed. Ecosystem modeling of Chesapeake Bay suggests that increased filtration by oysters could reduce eutrophication by lowering phytoplankton concentrations, which in turn would reduce the abundance of ctenophores and other gelatinous zooplankton (Ulanowicz and Tuttle 1992). Since gelatinous zooplankton are major predators of oyster larvae (Purcell et al. 1991), such an effect could be self-reinforcing.

Oyster reefs also interact directly with local hydrodynamic conditions, affecting currents, flow conditions, and sedimentation patterns (Lenihan 1999). They provide habitat as well as a refuge from both predation and poor water quality conditions for fish and decapods. Breitburg (1999) has shown that oyster reefs provide habitat for a variety of fish species, including resident species such as gobies, blennies, and toadfish (*Opsanus tau*), facultative species including black sea bass (*Centrpristes striata*), and transients such as juvenile winter (*Pleuronecets americanus*) and summer (*Paralichthes dentatus*) flounder, striped bass (*Morone saxatilis*), spot, and silversides (*Menidia menidia*). Examining fish and decapod distributions in subtidal habitats of North Inlet, South Carolina, Lehnert and Allen (2002) found distinctive resident and transient fish communities associated with shell bottoms. They further suggest that high densities of juvenile sea bass (*Centropristes* spp.), groupers (*Mycteroperca* spp.) and snappers (*Lutjanus* spp.) indicate that shell bottoms represent essential fish habitat for these species. Glancy et al. (2003) have compared fish and decapod communities of shallow-water habitats in a North Florida estuary and concluded that decapod fauna of oyster shell habitats are distinct from that of either seagrass or marsh-edge habitats. Posey et al. (1999) have experimentally shown that grass shrimp actively seek out oyster shells when predators are present, obtaining some refuge from predation.

POTENTIAL IMPACTS OF DREDGING IN SHALLOW-WATER HABITATS: The general impacts of dredging and dredged material disposal have been reviewed by Sherk and Cronin (1970), Windom (1976), Morton (1977), Guillory (1982), Allen and Hardy (1980), Diaz et al. (1982b), and most recently by Newell et al. (1998). All of these identify removal of habitat, burial, turbidity, altered current patterns, salinity intrusion, and decreased flushing in relatively deep areas as the most common effects associated with dredging projects. Although none of these potential issues are unique to shallow waters, they do suggest which critical shallow-water habitat functions are most at risk.

First, the direct physical impact of dredging entails removal of sediment potentially resulting in alteration of ambient sediment, water quality, and hydrodynamic conditions. Depending on the spatial scale involved, changes in bottom topography can have profound effects on benthic infauna. Dernie et al. (2003) have shown that a difference of only 10 cm in the amount of material removed from a Welsh sand flat resulted in a substantial decrease in the rate of recovery. Plots where 20 cm of sediment were removed required 208 days for the infaunal community to be reestablished, whereas plots where only 10 cm was removed recovered in 64 days. Of particular concern is the potential for reduced primary production by benthic microalgae in the dredged area due to decreased ambient light at depth. As has been discussed above, benthic microalgae are a critical component of estuarine primary production and directly support much of the secondary production of benthic invertebrates and fishes that use shallow-water habitats. Likewise, increasing depth contours by dredging may not only reduce value of the habitat as a predation refuge for estuarine-dependent fish and shellfish, but may provide a conduit for predators to reach areas which were previously inaccessible. In each of the cases described (benthic, microalgal, and fisheries impacts), the actual impact of the dredging operation will be a function of the spatial extent and degree to which the post-construction environment differs from pre-construction conditions.

Second, benthic habitats differ in their characteristic rate of recovery after disturbance. Newell et al. (1998) point out that benthic assemblages in low salinity (e.g., oligohaline) habitats recover faster than those in high salinity habitats (e.g., euhaline) and assemblages in fine-grained sediments recover faster than those in coarse-grained sediments. Such generalizations are not absolute, however, as shown by Ferns et al. (2000). They report that infauna of muddy-sand flats were far more impacted and required longer to recover from passage of tractor-powered cockle harvesters than those of sandy intertidal flats.

Finally, the ultimate use of the dredged area may result in persistent, deleterious impacts to surrounding habitats. Dredging of shallow water is generally performed to provide access for small boats. Vessel traffic in shallow navigation channels has the potential to create turbulence, resuspending bottom sediments, increasing turbidity, physically disrupting bottom communities, and producing wakes (or waves) that erode surrounding habitats. Beachler and Hill (2003) report that boats moving at "near plane" speeds create the greatest impact. They provide a model relating boat size and water depth to resuspension that may be of use in evaluating potential problems. Likewise, Maynord (2005) has produced a model relating wave height to the speed and size of small boats, which may be useful for evaluating wave-induced effects.

Repeated physical disturbance has the potential to result in decreased productivity and increased susceptibility of affected communities (Odum 1985, Gray 1989). Marinelli and Woodin (2002) demonstrated experimentally that disturbing the surface of soft sediments alters sediment chemistry, making it less attractive to recruiting infauna. If such disturbances are routine, the resulting communities will most likely be less abundant and less diverse than those of undisturbed habitats. Bishop (2004) and Bishop and Chapman (2004) have detected such impacts in intertidal infauna exposed to boat wakes.

Microalgal production is also sensitive to disturbance as shown by Emerson (1989). He found that production of microalgal communities (and higher trophic levels as well) is negatively correlated with the degree of wind stress, which represents both physical disruption and increased turbidity. Primary production of benthic microalgae and vascular plants (submerged aquatic vegetation or SAV) is closely tied to light availability, and increased turbidity generated by boating activity could inhibit this process (Dennison et al. 1993, Pinckney and Zingmark 1993, Barranguet et al. 1998, Kemp et al. 2004). Gucinski (1982) has examined the effects of boat prop-induced turbidity on photosynthesis in submerged aquatic vegetation and found that small boats traveling at high speeds in water depths of 1 m can resuspend sufficient amounts of sediment to reduce seagrass productivity. Boat traffic may also directly impinge on benthic microalgae or SAVs when boaters inadvertently stray out of the channel, creating prop scars, or disrupt the bottom when anchoring their vessels. Such direct impacts are already of concern for SAV's (Asplund and Cook 1997, Blackhurst and Cole 2000). Since most SAV's in Chesapeake Bay occur in depths less than 3 m (Orth and Moore 1988, Dennison et al. 1993), increased boat access could increase the numbers and frequency of such impacts.

Chronic resuspension of sediment may also represent a threat to nearby oyster reefs. Adult oysters are adapted to living in a turbid environment and are therefore relatively insensitive to moderate increases in turbidity or suspended sediments (Shumway 1996). Settling oyster larvae, however, are reputed to be sensitive to even very thin layers (a few millimeters) of sedimentation.

Boating activities also have the potential to interfere with nesting birds and other wildlife (York 1994). Shorebirds utilize shallow water for both feeding and nesting habitat. For instance, Burger (2003) has reported how slowing recreational boat traffic in Barnegat Bay, New Jersey decreased the number of "upflights" (a startle response) by the common tern (*Sterna hirundo*), resulting in more time spent nesting and increased reproductive success.

CONCLUSIONS: Shallow-water estuarine habitats represent ecologically important and generally under-valued natural resources. Historically, problems in assessing their value resided in the fact that the definition of the term "shallow" is largely context-sensitive. For instance, what is shallow water near the outer reaches of an estuary may represent relatively deep water in the upper estuary. Recent attempts to resolve this semantic difficulty have sparked renewed interest in determining the ecological functions of shallow waters and how they change along depth gradients. To date, however, relatively little quantitative information is available in the depth range of greatest interest, 0-2 m MLW.

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> Ray, G. L. (2005). "Ecological functions of shallow, unvegetated estuarine habitats and potential dredging impacts (with emphasis on Chesapeake Bay)," *WRAP Technical Notes Collection* (ERDC TN-WRAP-05-3), U. S. Army Engineer Research and Development Center, Vicksburg, MS. http://el.erdc.usace.army.mil/wrap

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Marine Pollution Bulletin 62 (2011) 520–524

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Effects of reclamation on macrobenthic assemblages in the coastline of the Arabian Gulf: A microcosm experimental approach

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ARTICLE INFO

Keywords:
Macrobenthic assemblages
Burial
Survival
Dredged material disposal
Vertical migration
Reclamation
Bahrain

ABSTRACT

Coastal reclamation and modifications are extensively carried out in Bahrain, which may physically smother the coastal and subtidal habitats resulting in changes to abundance and distribution of macrobenthic assemblages. A microcosm laboratory experiment using three common macrobenthic invertebrates from a proposed reclaimed coastal area was preformed to examine their responses to mud burial using marine sediment collected from a designated borrow area. Significant difference in numbers of survived organisms between control and experimental treatments with a survival percentage of 41.8% for all of the selected species was observed. The polychaete *Perinereis nuntia* showed the highest percentage of survival (57.1%) followed by the bivalve *Tellina valtonis* (42.3%) and the gastropod *Cerithidea cingulata* (24.0%). Quantifying species responses to sediment burial resulted from dredging and reclamation will aid in predicting the expected ecological impacts associated with coastal developments and subsequently minimizing these impacts and maintaining a sustainable use of coastal and marine ecosystems in the Arabian Gulf.

1. Introduction

The Arabian Gulf is considered among the highest anthropogenically impacted regions in the world (Halpern et al., 2008). Such impacts include dredging and reclamation, industrial and sewage effluents, hypersaline water discharge from desalination plants, and oil pollution (Sheppard et al., 2010). The coasts of the Arabian Gulf are undergoing rapid construction activities that often associated with intensive dredging and reclamation (Price and Sheppard, 1991). More than 40% of the coasts of the Arabian Gulf have been developed (Hamza and Munawar, 2009).

Likewise, Bahraini coastal and marine environments are the prime target for most of the major housing, recreational, and economic developments (Naser et al., 2008). Presently, reclamation activities in Bahrain resulted in adding around 91 km^2 representing an increase of 11% of the total land area (Zainal, 2009). Bahrain National Land Use Strategy 2030 recognizes reclamation as the major option for securing the future needs for land. Therefore, coastal environment in Bahrain will continue to be the major focus for developmental projects in the coming future.

Dredging and reclamation activities are typically associated with short and long term impacts that affect macrobenthic assemblages, water quality and fisheries, and cause coastal modifications (Marmoush, 1999; Airoldi and Beck, 2007). Macrobenthic assemblages are widely used as a tool to detect environmental impacts from several sources of stress and pollution (Naser, 2010a), including dredging and reclamation activities (Lu et al., 2002; French et al., 2004).

For the purposes of monitoring and environmental management, responses of macrobenthic assemblages to impacts of dredging and mud disposal have been studied in many parts of the world utilizing surveying and experimental approaches (Crus-Motta and Collins, 2004; Powilleit, et al., 2006; Wilber et al., 2007; Whomorsley et al., 2010; Ware et al., 2010). Conversely, studies investigating the anthropogenic effects on macrobenthic assemblages have been limited in the Arabian Gulf, and responses of these assemblages to the expected impacts that associated with dredging and reclamation related to major projects are rarely quantified in environmental impact assessment reports (Naser, 2010b).

Giving the increasing coastal modifications in Bahrain, measures to detect and quantify disturbances affecting benthic ecosystems are critically required. One approach to investigating the expected impacts of reclamation on macrobenthic assemblages is by utilizing microcosm experiments. Even though the relevance of such experiments to the complexity of natural ecological communities has been subjected to debate (Carpenter, 1996; Balvanera et al., 2006), they can provide quantified evidences of the response of organisms toward environmental impacts (Stark, 1998; Solomon and Sibley, 2002).

Using microcosm experimental approach to quantify the ecological impacts associated with coastal developments may

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doi:10.1016/j.marpolbul.2010.11.032

consequently minimize these impacts and maintain sustainable use of coastal and marine ecosystems. This study, therefore, applies a microcosm experimental approach to investigate the effects of mud burial on key species of macrobenthic assemblages inhabiting a shallow coastal area in Bahrain that proposed to be reclaimed using marine sediment collected from a designated borrow area.

2. Materials and methods

2.1. Macrobenthic assemblages in the reclamation area

The selected intertidal area (N 26°09′29.97″ and E 50°38′24.59.26″) is proposed to be reclaimed for housing development (Fig. 1). This area (ca. 5 km^2) is shallow with a depth ranges from 0 to 4 m. The substrate is predominately covered by thin layer of fine and medium sands (average sand depth = 12.1 ± 5.2 cm). Sediment samples were collected form the inertial area from a 0.1 m^2 quadrat to an approximate depth of 10 cm. Three transects with three stations at the upper, middle and lower intertidal zones were established. Three replicate samples were collected at each station. Faunal samples were sieved in situ through a 1.0 mm mesh using seawater, fixed in 4% formalin, stained with drops of rose Bengal and subsequently preserved in 70% ethyl alcohol. Organisms were sorted according to their taxonomic groups, counted and identified to the lowest possible taxonomic level using relevant identification guides (Green, 1994; Bosch et al., 1995; Richmond, 2002; Wehe and Fiege, 2002).

A total of 628 individual organisms belonging to 43 species were recorded in the study area. Polychaeta was the best represented taxon with 22 species followed by Mollusca and Crustacea with nine and seven species, respectively. The remaining five species were represented by Echinodermata, Sipuncula, and Chordata with one, three, and one species, respectively. Polychaetes were also the most abundant among the major taxonomic groups representing 79.0% of the total population followed by crustaceans 11.9%, molluscs 7.8%, and the remaining of the groups with 1.3%. The abundance and spatial distribution of the assemblages inhabiting the reclamation area were used to select species for the laboratory microcosm experiment.

2.2. Taxa used in the microcosm experiment

Selecting indicator species to detect environmental condition or impact is based on ecological criteria as well as socio-economic qualities such as the relevance to policy and management needs (Goodsell et al., 2009; Marques et al., 2009). The taxa used in the microcosm experiment are common in the study area, and were selected based on their higher levels of abundance.

Two species of molluscs were used; the bivalve *Tellina valtonis*, which dominated the molluscs population in the reclamation area with 67%, and the gastropod *Cerithidea cingulata*, which accounted for 13% of the molluscs. *T. valtonis* is a filter feeder that dominates fine sand substrates, while the tropical mud snail *C. cingulata* is a grazer that commonly found in mudflats around Bahrain (Al-Sayed et al., 2008). The other taxon used in this study was the polychaete *Perinereis nuntia,* which accounted for 54% the polychaetes population. *P. nuntia* is an omnivore that mainly feeds on drift algae (Rouse and Pleijel, 2001).

2.3. Experimental design and running the microcosm

Sediment samples were collected from the proposed reclaimed area and a subtidal borrow area within the territorial waters of Bahrain (N 26°14′13.44″ and E 50°49′59.26″) that was designated for dredging the material to be used in the reclamation process. Sediments from reclamation and borrow areas were collected in March 2010 using a corer and Van Veen grab intertidally and subtidally, respectively.

Sediment samples were kept at 0 °C to prevent decomposition until treatment phase. Portions of the collected sediments were used to analyse the sediment grain size, which involved sieving 50 g of homogenized sediment on a mechanical shaker (KARL KOB) through six sieves (mesh sizes from 0.038–2 mm) and obtaining weight of sediment fraction in each sieve. The percentage of organic content of both types of the sediment was obtained by incinerating a known weight at a temperature of 450 °C for 12 h.

The microcosm was constructed using ten cylindrical PVC tubes (inner diameter 6.5 cm, length 50 cm). These tubes were sealed from one end using removable plastic stoppers and equally divided

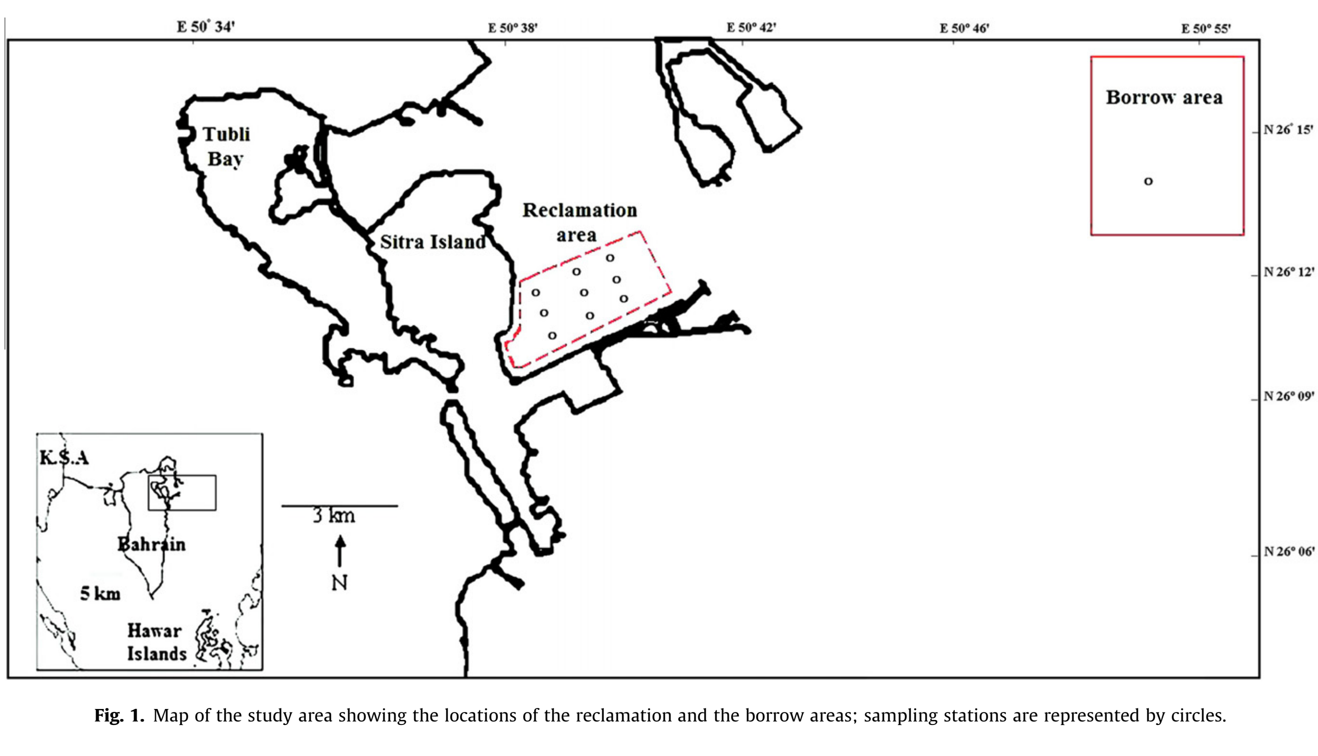

Fig. 1. Map of the study area showing the locations of the reclamation and the borrow areas; sampling stations are represented by circles.

between experimental and control ones (5-experimental and 5-control).

The selected organisms were collected from the reclamation area by both digging and sieving. Organisms collected in the field were transported back to the laboratory within 2 h of collection. Only individuals in good condition were used in the microcosm. Organisms were immediately distributed over the tubes that contain 15 cm of the intertidal sediment and allowed to acclimate for 48 h in a controlled temperature (22 °C) and under aerated filtered seawater. Each tube contains six individuals form each selected species (total of organisms in experimental and control tubes = 180). Another 15 cm of the defaunated sediment that collected from the borrow area were added to the experimental tubes, and both tube sets were kept for 14 days. After that period, sediment from each tube was carefully released as a cylinder and sliced each 3 cm, which was resulted in five slices for controls and 10 slices for the experimental ones. Each slice was examined for the presence of the organisms and subsequently sieved through 0.4 mm mesh to retain unobserved organisms.

2.4. Statistical analysis

The response of the selected species to sediment burial was statistically analyzed by cross-tabulation using Chi square test with null hypothesis that mud burial has no effect on the selected species. The composition of the selected faunal species in both experimental and control tubes was analyzed by non-metric multidimensional scaling (MDS) based on the Bray–Curtis dissimilarity coefficient using square root transformed data (Clarke and Warwick, 2001). Statistical tests were performed using SPSS v15 and PRIMER® *v* 6 (Clarke and Glorley, 2006) Statistical packages.

3. Results

3.1. Sediment characteristics

Mean grain size analyses for sediments collected from reclamation and borrow areas indicated that they are primarily composed of fine to coarse sand ($\Phi = 2.69 \pm 0.28$) and very fine sand ($\Phi = 3.13 \pm 0.12$), respectively. The percentages of organic content were $4.3 \pm 2.1\%$ and $7.4 \pm 1.9\%$ for reclamation and borrow areas, respectively.

3.2. Survival and vertical burrowing of the selected species

Statistical analysis showed significant differences in numbers of survived individuals between control and experimental treatments with a survival percentage of 41.8% for all of the selected species ($\chi^2 = 21.78$, df = 1, $P = 0.002$). The multidimensional scaling (MDS) revealed a clear distinction between organisms in the control and those in the experimental tubes (Fig. 2). However, variations in percentages of survival were observed among the tested faunal species. *P. nuntia* showed the highest percentage of survival (57.1%) followed by *T. valtonis* (42.3%) and *C. cingulata* (24.0%).

P. nuntia showed a high vertical burrowing potential after burial with 43.8% of the survived individuals reaching the top 3 cm followed by *T. valtonis* as 54.5% of the survived individuals moved-up nearly 9 cm. Conversely, vertical burrowing was limited by *C. cingulata* (Fig. 3).

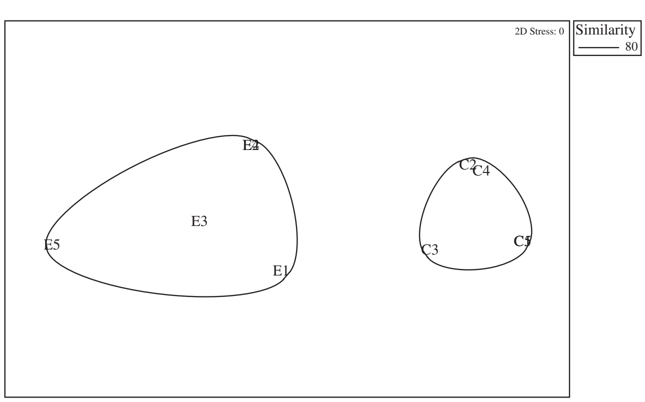

Fig. 2. MDS plot using Bray–Curtis similarity coefficient shows a clear segregation between assemblages in control and experimental tubes. Resemblance matrix was clustered to generate similarity of 80% of the MDS. C = Control and E = Experimental; numbers are correspond to tubes.

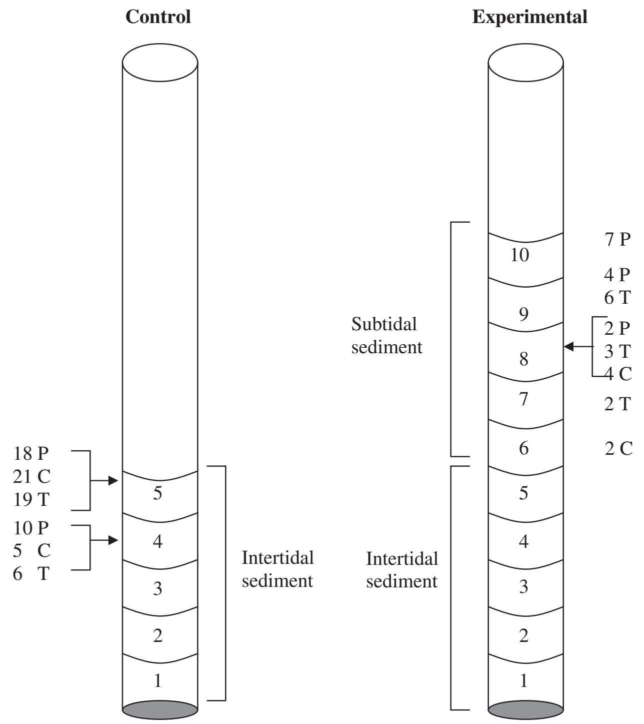

Fig. 3. Total numbers of survived organisms pooled from the control and experimental tubes and their vertical migration at the end of the experiment. *Perinereis nuntia* showed highest potential of vertical burrowing with 43.8% of the survived individuals reaching the top 3 cm. P = *Perinereis nuntia*, T = *Tellina valtonis*, C = *Cerithidea cingulata*.

4. Discussion

Dredging and reclamation processes are regularly carried out in Bahrain to meet the demand for rapid coastal development. Sand and mud characterized by lower levels of biodiversity and abundance are extracted from borrow areas within the territorial waters of Bahrain then dumped into coastal and subtidal areas characterized by high levels of biodiversity and productivity (Naser, 2010a). Deposition of dredged material that associated with reclamation process results in physically smothering the coastal and subtidal habitats and deoxygenating the underlying sediments. These environmental effects may cause changes in the abundance and distribution of benthic organisms within the reclaimed area (Smith and Rule, 2001; Lu et al., 2002).

Macrobenthic assemblages do not respond in a predictive manner to dredged material disposal (Harvey et al., 1998), which could be attributed to the different biological, physical, and chemical components of dredged material and receiving area. Microcosm experiments can contribute into providing rapid quantified evidences of expected ecological impacts associated with reclamation. Such evidences could be used to inform decision-makers with the environmental consequences of reclamation before authorization for coastal developments is granted. Additionally, microcosm experiments may overcome obstacles associated with long-term environmental studies by reducing time and cost and minimizing disturbance of sensitive areas with legal obligations such as conservation designation. Therefore, microcosm experiments could be utilized to achieve a balance between impacts prediction of proposed developments and maintaining legal, time and cost obligations.

The current microcosm experiment revealed that the selected species were affected differently by the disposed mud, which could be attributed to the behaviour and physiology of the buried organism (Maurer et al., 1981). Several studies indicated that large and highly motile polychaetes showed the highest tolerance to sediment disposal (McGrorty and Reading, 1984; Chou et al., 2004; Naser, 2010a). Similarly, the actively motile *P. nuntia* showed the highest percentage of survival and vertical migration, suggesting that motile polychaetes might be more tolerant to sediment dumping than molluscs.

Cerithidea species are considered surface organisms that widely speared on sand and muddy shores (Houbrick, 1984) this may explain the high percentage of mortality in *C. cingulata* after burial in the current microcosm. All of the selected species are habitat-specialists and the levels of their abundance may affect other organisms such as wader birds that feed on them.

For any laboratory microcosm, it is difficult to simulate the complex temporal and spatial environmental conditions for natural coastal and marine ecosystems. Therefore, such experiments have limited capacity to generalize about environmental effects of reclamation on macrobenthic assemblages in the filed (Schratzberger et al., 2000; Mayer-Pinto et al., 2010). Nonetheless, the current microcosm attempted to initially quantify the responses of macrobenthic assemblages to ongoing reclamation activities in Bahrain. Although macrobenthos in the Arabian Gulf are characterized by high levels of biodiversity (Al-Yamani et al., 2009), they are distinguished by low species richness due to the harsh environmental conditions (Basson et al., 1977). Therefore, reclamation effect could arguably be critical for macrobenthos inhabiting the naturally stressed marine environment of the Arabian Gulf. Consequently, estimation the loss of macrobenthic assemblages as a result of dumping and physical alternation of the coasts of Arabian Gulf is considered to be a priority research objective (Sheppard et al., 2010).

Quantifying the environmental impacts associated with dredging and reclamation will aid in predicting how macrobenthic assemblages could change in the coastal and marine environments in Bahrain, which has several benefits and applications in the context of environmental and monitoring studies. These include estimating ecological compensation, validating the prediction of impacts and verifying the effectiveness of mitigation measures, and planning for conservation and management of coastal and marine habitats.

This study concluded that investigating species responses to sediment burial may provide a better understanding of the effects of reclamation on macrobenthic assemblages in the coastal and marine environments of the Arabian Gulf.

Acknowledgments

Thanks are due to the staff at the Department of Biology, College of Science, University of Bahrain for their logistical and technical help. Assistance provided by Mr. Khalil Al-Wedaie during sampling is gratefully appreciated.

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Marine Pollution Bulletin 52 (2006) 1553–1572

MARINE POLLUTION BULLETIN

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Review

Environmental impacts of dredging on seagrasses: A review

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Abstract

Main potential impacts on seagrasses from dredging and sand mining include physical removal and/or burial of vegetation and effects of increased turbidity and sedimentation. For seagrasses, the critical threshold for turbidity and sedimentation, as well as the duration that seagrasses can survive periods of high turbidity or excessive sedimentation vary greatly among species. Larger, slow-growing climax species with substantial carbohydrate reserves show greater resilience to such events than smaller opportunistic species, but the latter display much faster post-dredging recovery when water quality conditions return to their original state. A review of 45 case studies worldwide, accounting for a total loss of 21,023 ha of seagrass vegetation due to dredging, is indicative of the scale of the impact of dredging on seagrasses. In recent years, tighter control in the form of strict regulations, proper enforcement and monitoring, and mitigating measures together with proper impact assessment and development of new environmental dredging techniques help to prevent or minimize adverse impacts on seagrasses. Costs of such measures are difficult to estimate, but seem negligible in comparison with costs of seagrass restoration programmes, which are typically small-scale in approach and often have limited success. Copying of dredging criteria used in one geographic area to a dredging operation in another may in some cases lead to exaggerated limitations resulting in unnecessary costs and delays in dredging operations, or in other cases could prove damaging to seagrass ecosystems. Meaningful criteria to limit the extent and turbidity of dredging plumes and their effects will always require site-specific evaluations and should take into account the natural variability of local background turbidity.

Keywords: Critical thresholds; Dredging impacts; Mitigation; Review; Seagrasses; Turbidity

1. Introduction

Dredging is required in many ports of the world, to deepen and maintain navigation channels and harbour entrances. Elsewhere, commercial sand mining and extraction of sand and gravel from borrowing areas is meeting an ever-increasing demand for sand for construction and land reclamation. Excavation, transportation and disposal of soft-bottom material may, however, lead to various adverse impacts on the marine environment (USACE, 1983; ABP Research, 1999). Such impacts can be especially significant when dredging or disposal is done in the vicinity of sensitive marine environments, such as coral reefs and seagrass beds.

Seagrass beds, covering about 0.1–0.2% of the global ocean floor, are highly productive ecosystems which fulfill a key role in the coastal zone (Duarte, 2002). Important ecological and economic functions of seagrass beds have been widely acknowledged, notably their importance to fisheries (Bell and Pollard, 1989; Jackson et al., 2001) and their role in preventing coastal erosion and siltation of coral reefs (Scoffin, 1979; Fonseca and Fisher, 1986; Fonseca, 1989). Conservative estimates of the value of ecosystem services provided by seagrass beds are in the order of 19,000 US$ ha^{-1} yr^{-1} (Costanza et al., 1997).

Globally, the estimated loss of seagrass from direct and indirect human impacts is updated to be 33,000 km^2 over the last two decades, based on an extrapolation of known losses (Short and Wyllie-Echeverria, 2000) and a new global seagrass area estimate of 177,000 km^2 (Green and Short, 2003). The primary cause of seagrass degradation

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0025-326X/$ - see front matter © 2006 Elsevier Ltd. All rights reserved.
doi:10.1016/j.marpolbul.2006.09.006

and loss globally is reduction in water clarity, both from increased turbidity and increased nutrient loading (Walker and McComb, 1992; Duarte, 2002; Short, 2003). In many cases, dredging operations have directly or indirectly contributed to loss of seagrass vegetation. For example, combined impacts of increased turbidity and physical removal or burial during dredging, and eutrophication from nutrients in domestic and industrial discharges caused the loss of approximately 81% of the seagrasses in Tampa Bay, Florida (Lewis, 1976; Lewis et al., 1985).

Some seagrass species appear more sensitive than others to increases in turbidity or sedimentation that are commonly associated with dredging operations. Their response to such increases may depend on typical local conditions and vary between seasons. In general, the impact from dredging and sand mining on seagrass ecosystems is complex and far from fully understood, despite various research efforts. Initial investigation by the authors has shown that there is an extensive body of experience to learn from. This experience lies with contractors, in Environmental Impact Assessment (EIA) reports, monitoring data and scientific literature derived from field-based and laboratory studies on potential direct and indirect effects of dredging.

This paper presents a worldwide review of documented cases of dredging and sand mining operations in or near seagrass meadows. The scale of such operations and environmental impacts thereof, as well as restrictions and mitigating measures put into place to minimize these impacts will be discussed. Where appropriate, these findings will be illustrated with case studies. The focus of this review is limited to dredging for the purpose of deepening of ports and navigation channels as well as for extraction and mining of sediments for construction and land reclamation schemes. Other forms of dredging, such as hydraulic clam dredging and cockle fishing, or damage to seagrass beds by boat groundings, propellor scarring and anchoring, were not considered. Information sources for the review included peer-reviewed scientific literature, gray literature in the form of EIA-, consultancy and technical reports, and additional information obtained from the internet and email responses to general requests placed on internet-based research discussion lists.

2. Environmental impacts of dredging

Potential effects of dredging on the marine environment include effects of the dredging process (i.e. the removal of substratum from the seafloor) as well as effects caused by the process of disposal. Dredged material may come into suspension during dredging itself as a result of disturbance of the substratum, but also during transport to the surface, overflow from barges or leakage of pipelines, during transport between dredging and disposal sites, and during disposal of dredged material (Jensen and Mogensen, 2000).

Dredging may affect the physical environment by changing the bathymetry, altering current velocities and wave conditions (Jensen and Mogensen, 2000) which affect the sedimentary regime and may cause erosion under seagrass beds (MacInnis-Ng, 2003).

Dredging and disposal of dredged material can lead to a temporary decrease in water transparency, increased concentrations of suspended matter, and increased rates of sedimentation. In the case of contaminated sediment or sediments with high contents of organic matter, dredging and resuspension may also lead to effects on water quality by the release of contaminants (e.g. Filho et al., 2004), an increase in nutrients concentrations and reduced dissolved oxygen in the water column.

Physical removal of substratum and associated plants and animals from the seabed, and burial due to subsequent deposition of material are the most likely direct effects of dredging and reclamation projects (Newell et al., 1998). New habitats may also be created as a result of the operation, either directly in the dredged area or by introduction of new habitats on the slopes of a reclaimed area (e.g. hard substratum in the form of breakwaters and revetments).

Other direct effects may be caused by enhanced turbidity and sedimentation as a result of dredging and disposal operations. The effect of turbidity on seagrass ecosystems is two-fold. Light attenuation by suspended material affects the amount of light available to the seagrass plants and associated epiphytes, microphytobenthos and macroalgae. Depending on the depth at which these organisms occur, high turbidity can cause a significant reduction in light availability leading to sub-lethal effects or death. High levels of suspended material can lead to reduced vitality or death in benthic fauna associated with the seagrass beds through clogging of their feeding mechanisms (cilia and siphons) and smothering, especially in filter-feeding organisms such as mussels, oysters and other bivalves. To capture both effects of turbidity, critical thresholds for turbidity should therefore ideally be determined in terms of light availability at the bottom (in % of surface irradiance) as well as in concentration of total suspended solids (in mg/l).

Increases in turbidity can also be caused by algal blooms, sewage discharge, bio-fouling of turbidity sensors etc. Turbidity should therefore not only be expressed in terms of a reduction of light availability as the sole measure of water quality affected by dredging works, but preferably be accompanied by investigations of the suspended solid concentrations (Bogers and Gardner, 2004).

Turbidity changes induced by dredging will only result in adverse environmental effects when the turbidity generated is significantly larger than the natural variation of turbidity and sedimentation rates in the area (Stern and Stickle, 1978; Orpin et al., 2004). Such natural variability can sometimes be substantial and may be caused by factors such as storms, wind-induced wave actions, river discharges and other local perturbations. Dredging activities often generate no more increased suspended sediments than commercial shipping operations, bottom fishing or severe storms (Pennekamp et al., 1996).

The degree of adverse environmental impacts caused by dredging and disposal depends on the quantity, frequency and duration of dredging, methodology of dredging and disposal, physical dimensions and water depth of the dredging location, grain-size composition, density and degree of contamination of the dredged material, background water quality (especially suspended matter and turbidity), seasonal variations in weather conditions (especially wind and waves), and proximity/distance of ecologically sensitive or economically important areas or species relative to the location of the dredging or disposal site (Pennekamp et al., 1996). Depending on these factors, there can be considerable spatial and temporal variation in effects. In some cases, adverse impacts of dredging activities are limited to a relatively small area and of relatively short duration. Other (more large-scale) dredging or sand mining operations, which stretch out over several years and cover many square kilometres, can have major adverse environmental impacts (Lewis, 1976).

In summary, main potential impacts from dredging on seagrasses include physical removal or burial of vegetation at the dredging/disposal site, and increased turbidity (light reduction) and increased sedimentation in adjacent seagrass meadows. In addition, temporarily reduced dissolved oxygen concentration, release of nutrients and pollutants from (contaminated) sediments, and hydrographic changes may also occur and have adverse (indirect) effects on the seagrass ecosystem.

3. Critical thresholds of seagrasses for turbidity

Light is one of the key environmental resources imperative for the growth and survival of seagrasses (Hemminga and Duarte, 2000). Water transparency (which determines depth-penetration of photosynthetically active radiation of sunlight) is the primary factor determining the maximum depth at which seagrasses can occur. Reduction in light due to turbidity has been identified as a major cause of loss of seagrasses worldwide (Shepherd et al., 1989; Green and Short, 2003). The amount of light that reaches a seagrass leaf is determined by the natural water colour, concentration of suspended solids, phytoplankton concentration and epiphyte cover of the leaf (Dennison, 1991; Batiuk et al., 2000).

There are various reports of sublethal and lethal effects on seagrass meadows due to prolonged exposure to high turbidity and siltation associated with dredging activities (Caldwell, 1985; Gaby et al., 1986; Onuf, 1994; Gordon et al., 1996; Chesire et al., 2002; Sabol et al., 2005). Indicators of light stress in seagrasses may include decreases in below-ground biomass and carbohydrate contents of rhizomes, tissue nutrient contents, Chl-a contents of leaves and various photosynthetic growth parameters (Coles and McKenzie, 2004).

There is a considerable range of values (2.5–37% of SI) reported in the literature for the minimum light requirements of seagrasses, varying between different seagrass species as well as within a single seagrass species (Table 1). The order of magnitude of this variation is similar between species and within species (Fig. 1). Minimum light requirements of most seagrass species seem to vary between 15% and 25% of SI (means of reported values per species), but for some species (*Cymodocea nodosa*, most *Halophila* spp. and some *Posidonia* spp.) minimum light requirements as low as 3–8% of SI have been reported (Table 1, Fig. 1).

The variation in minimum light requirements reported in literature is in part caused by differences in the methodologies used to derive these values. Methodologies range from physiological studies of photosynthesis/irradiance relationships, field observations of maximum depth of seagrass colonization, and experimental manipulation of light levels during growth studies, to statistical models (Batiuk et al., 2000). Studies and methods further differ in the degree to which attenuation by epiphyte cover of seagrass leaves, natural water colour, seasonal variation, above-/below-ground biomass ratios, environmental factors other than light, and sublethal effects have been taken into account. Mesocosm experiments have clearly shown effects of shading on plant architecture, biomass partitioning, lateral shoot development and flowering intensity in eelgrass (Ochieng et al., 2004).

Whilst minimum light requirements are important, it is an oversimplification to assume that light attenuation alone determines plant response to increased turbidity. Also of importance is the length of time that different species can survive at low light levels. Temporary fluctuations in turbidity may be accommodated by the plant depending on the nature of the species and the period of sub-optimal light (Westphalen et al., 2004). Laboratory experiments have shown that some seagrasses can survive in light intensities below their minimum requirements for periods ranging from a few weeks to several months (Table 2) (Backman and Barilotti, 1976; Bulthuis, 1983; Gordon et al., 1994; Czerny and Dunton, 1995; Longstaff et al., 1999). Widespread seagrass mortality was observed in Chesapeake Bay (USA) following a month-long (seasonal) pulse of increased turbidity (Moore et al., 1997).

The survival period of seagrass below its minimum light requirement is shorter in smaller species with low carbohydrate storage capacity than in larger species (Longstaff et al., 1999; Peralta et al., 2002). Work by Chesire et al. (2002) indicates that *Posidonia sinuosa* is able to survive longer at sub-compensation light levels than *Zostera tasmanica*, which in turn survives slightly longer than *Z. marina*, while *Halophila ovalis* copes with sub-minimal light for the shortest period. Whilst these results used different methodologies and measurements to determine survival, it is clear that species with larger below-ground biomass are better adapted to longer periods of sub-minimal light.

4. Critical thresholds of seagrasses for sedimentation

Several studies have documented deterioration of seagrass meadows by smothering due to excessive

Table 1
Critical threshold of seagrasses for light availability ('minimum light requirements' expressed as % of surface irradiance SI)

Species	Location	% SI	Reference
Amphibolis antarctica	Waterloo Bay, Australia	24,7	Duarte (1991)
Cymodocea nodosa	Malta	7,3	Drew (1978)
Cymodocea nodosa	Ebro Delta, Spain	10,2	Duarte (1991)
Halodule wrightii	Laguna Madre, Texas, USA	5	Dunton and Tomasko (1991)
Halodule wrightii	Alabama, USA	14	Shafer (1999)
Halodule wrightii	Texas, USA	16	Czerny and Dunton (1995)
Halodule wrightii	Texas, USA	16	Onuf (1996)
Halodule wrightii	Florida, USA	17,2	Dennison et al. (1993)
Halodule wrightii	Texas, USA	17,5	Onuf (1991)
Halodule wrightii	Texas, USA	18	Dunton (1994)
Halodule wrightii	Texas, USA	18	Dunton and Tomasko (1994)
Halodule wrightii	Indian River Lagoon, Florida, USA	29,5	Beal and Schmit (2000)
Halodule wrightii	Indian River Lagoon, Florida, USA	30,5	Kenworthy and Fonseca (1996)
Halophila decipiens	Hobe Sound, Florida, USA	2,5	Dennison (1987)
Halophila decipiens	St.Croix, Caribbean	4,4	Williams and Dennison, 1990
Halophila decipiens	Northwest Cuba	8,8	Duarte (1991)
Halophila engelmanni	Northwest Cuba	23,7	Duarte (1991)
Halophila ovalis	Zanzibar, Tanzania	16	Schwarz et al. (2000)
Halophila spp.	Sub tropical seas	5	Dennison et al. (1993)
Halophila stlpulacea	GulfofEilat, Red Sea	3	Beer and Waisel (1982)
Heterozostera tasmanica	Spencer Gulf, Australia	4,4	Duarte (1991)
Heterozostera tasmanica	Victoria, Australia	5	Bulthuis (1983)
Heterozostera tasmanica	Australia	9	Bulthuis and Woelkerling (1983)
Heterozostera tasmanica	Chile	17,4	Duarte (1991)
Heterozostera tasmanica	Waterloo Bay, Australia	20,2	Duarte (1991)
Posidonia angustifolia	Waterloo Bay, Australia	24,7	Duarte (1991)
Posidonia australis	Australia	10	Fitzpatrick and Kirkman (1995)
Posidonia coriacea	Adelaide coast, Australia	8	Westphalen et al. (2004)
Posidonia oceanica	Medas Island, Spain	7,8	Duarte (1991)
Posidonia oceanica	Malta	9,2	Drew (1978)
Posidonia oceanica	Corsica, France	10–16%	Dalla Via et al. (1998), Ruiz and Romero (2003)
Posidonia ostenfeldii	Waterloo Bay, Australia	24,7	Duarte (1991)
Posidonia sinuosa	Australia	20	Gordon et al. (1994)
Posidonia sinuosa	Waterloo Bay, Australia	24,7	Duarte (1991)
Ruppiamaritima	Brazil	8,2	Duarte (1991)
Ruppia maritima	Australian estuary	28	Congdon and McComb (1979)
Ruppia megacarpa	Western Australia	24	Carruthers et al. (1999)
Syringodium filiforme	Florida, USA	17,2	Dennison et al. (1993)
Syringodium filiforme	Florida, USA	18,3	Duarte (1991)
Syringodium filiforme	Northwest Cuba	19,2	Duarte (1991)
Syringodium filiforme	Indian River Lagoon, Florida, USA	30,5	Kenworthy and Fonseca (1996)
Thalassia testudinum	Florida, USA	10	Fourqurean et al. (1999)
Thalassia testudinum	Texas, USA	14	Czermy and Dunton (1995)
Thalassia testudinum	Texas, USA	14	Lee and Dunton (1997)
Thalassia testudinum	Florida, USA	15	Fourqurean and Zieman (1991)
Thalassia testudinum	Florida, USA	15,3	Duarte (1991)
Thalassia testudinum	Gulf of Mexico	20	Iverson and Bittaker (1986)
Thalassia testudinum	Florida Bay, USA	20	Stumpf et al. (1999)
Thalassia testudinum	Tampa Bay, Florida, USA	22,2	Dixon (2000)
Thalassia testudinum	Northwest Cuba	23,5	Duarte (1991)
Thalassia testudinum	Puerto Rico	24,4	Vicente and Rivera (1982)
Zostera capricomi	Moreton Bay, Australia	30	Longstaff et al. (1999)
Zostera capricomi	Moreton Bay, Australia	30	Abal and Dennison (1996)
Zostera marina	New Hampshire, USA	11	Short et al. (1995)
Zostera marina	Japan	18,2	Duarte (1991)
Zostera marina	Woods Hole, USA	18,6	Dennison (1987)
Zostera marina	Roskilde, Denmark	19,4	Borum (1983)
Zostera marina	Chesapeake Bay, USA	20	Dennison et al. (1993)
Zostera marina	Long Island Sound, USA	20	Burkholder and Doheny (1968)
Zostera marina	Kattegat, Denmark	20,1	Ostenfeld (1908)
Zostera marina	Denmark	20,6	Duarte (1991)
Zostera marina	York River, VA (USA)	25	Moore (1991)
Zostera marina	Netherlands	29,4	Duarte (1991)
Zostera marina	Long Island Sound, USA	35,7	Koch and Beer (1996)

Table 1 (*continued*)

Species	Location	% SI	Reference
Zostera marina	California, USA	37	Backman and Barilotti (1976)
Zostera noltii	Spain	2	Peralta et al. (2002)

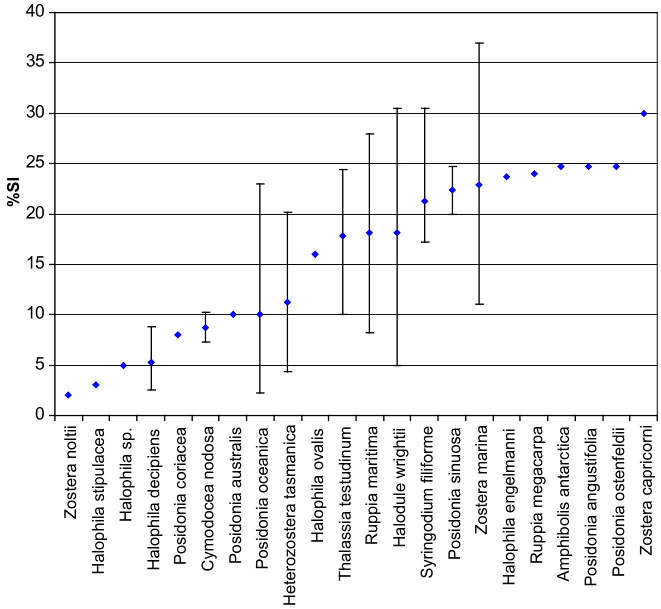

Fig. 1. Range of critical threshold values for light availability (as % surface irradiance SI) reported in the literature for various seagrass species.

sedimentation. Consequences of enhanced sedimentation for seagrass plants depend on several factors such as depth of burial and life history of the species involved (Duarte et al., 1997). Seagrass species that develop vertical shoots (e.g. *Cymodocea*, *Thalassia*, *Thalassodendron*) may respond to fluctuations in sediment depth by modifying their vertical growth to relocate their leaf-producing meristems closer to the new sediment level, but there are limits to the level of sedimentation seagrasses can tolerate (Marba and Duarte, 1994).

Vermaat et al. (1997) reported sedimentation rates of 10–13 cm yr^{-1} as maximum threshold value of what seagrasses in the Philippines and Spain can survive. Manzanera et al. (1995) reported significant mortality of shoots of the seagrass *Posidonia oceanica* in response to experimental oversedimentation, even at moderate burial (ca. 5 cm). Mills and Fonseca (2003) observed >50% mortality of *Zostera marina* in field burial treatments of 4 cm (corresponding with 25% of plant height) for 24 days. Plants responded similarly to burial in either sand or silt. Plants buried 75% or more of their height (16 cm) experienced 100% mortality.

An overview of values reported in literature for maximum allowable sedimentation rates for seagrasses is presented in Table 3.

In general, it is difficult to separate the effects of turbidity and sedimentation in field studies. Settlement of suspended material on leaf blades of seagrasses may interfere significantly with photosynthesis, and appears especially significant in low wave energy environments where fine sediments are present and can settle out (Shepherd et al., 1989). The impact of sedimentation is often increased where epiphytes are abundant on seagrass leaves (for instance under nutrient enriched conditions) because epiphitized leaf blades collect a greater amount of sediment. In the case of eelgrass (*Zostera marina*) blades and epiphytes then appear dull brown coated with a fine layer of sediment, and they often sink to the bottom (Short et al., 1989).

An indication of the duration that seagrasses can tolerate high rates of sedimentation was revealed by field experiments in Spain. Artificial burial of the seagrass *Posidonia oceanica* with as much as 15 cm of sediment caused 100% mortality after 200–300 days (Manzanera et al., 1995). Sudden burial of *Cymodocea nodosa* with 5 cm of sediment resulted in 90% mortality after 35 days, although some individual shoots of this species were able to survive burial as great as 6 cm (Marba and Duarte, 1994).

Experimental burial of a mixed species seagrass meadow in the Philippines with varying amounts of sediment resulted in major differences in species response (Duarte et al., 1997). *Thalassia hemprichii* and *Cymodocea rotundata* showed a sharp decline in shoot density even at moderate burial treatment with still no recovery 2 months after

Table 2
Duration of time that seagrass species can survive in light intensities below their minimum light requirements

Species	Light availability	Period survived (month)	Reference
Halodule pinifolia	0	3–4	Longstaff and Dennison (1999)
Halodule wrightii	13–15% SI	9	Czerny and Dunton (1995)
Halophila ovalis	0	1	Longstaff et al. (1999)
Heterozostera tasmanica	9% SI	10	Bulthuis (1983)
Heterozostera tasmanica	2% SI	2–4	Bulthuis (1983)
Posidonia sinuosa	12% ambient	24	Gordon et al. (1994)
Thalassia testudinum	10% SI	11	Czerny and Dunton (1995)
Zostera capricomi	5% SI	1	Grice et al. (1996)
Zostera noltii	<2% SI	0.5	Peralta et al. (2002)

Table 3
Critical thresholds of seagrasses for sedimentation (cm/year)

Species	Location	Sedimentation (cm/yr)	Reference
Cymodocea nodosa	Mediterranean (Spain)	5	Marba and Duarte (1994)
Cymodocea rotundata	Philippines	1.5	Vermaat et al. (1997)
Cymodocea serrulata	Philippines	13	Vermaat et al. (1997)
Enhalus acoroides	Philippines	10	Vermaat et al. (1997)
Halophila ovalis	Philippines	2	Vermaat et al. (1997)
Posidonia oceanica	Mediterranean (Spain)	5	Manzanera et al. (1995)
Zostera noltii	Mediterranean (Spain)	2	Vermaat et al. (1997)

burial. *Halodule uninervis*, *Syringodium isoetifolium* and *Cymodocea serrulata* showed an initial decline in shoot density followed by recovery. *Enhalus acoroides* maintained shoot density at all burial treatments and only showed some evidence of decline by the end of the experiment. *Halophila ovalis* showed an opportunistic growth in plots receiving 4–8 cm of sediment, reaching shoot densities well in excess to those of control plots (Duarte et al., 1997).

Sediment conditions (silt and clay content, organic matter and sulfide concentration) can be an important factor limiting seagrass distribution (Koch, 2001), as supported by observations in Southeast Asia where both species diversity and leaf biomass of seagrass communities declined sharply when the silt and clay content of the sediment exceeded 15% (Terrados et al., 1998). Under conditions of high light availability, however, major changes in sediment conditions associated with siltation may not negatively affect seagrass plants but instead enhance their growth by increasing the availability of nutrients, as revealed by recent experiments with *Cymodocea rotundata* in the Philippines (Halun et al., 2002).

Clarke (1987) undertook a series of field-based burial experiments on seagrasses along the Adelaide coast, which demonstrated that, as long as the sediments remained aerobic, *Amphibolis* plants were unaffected in terms of growth rate by burial up to 10 cm of sediment. This was contrasted with *Posidonia angustifolia*, which demonstrated an inverse relationship between growth and depth of burial, unless conditions were anaerobic, causing mortality within 2 weeks (Clarke, 1987). *Zostera tasmanica* appears to be particularly vulnerable to high deposition environments (esp. in the intertidal) as the leaves are quickly coated with sediment (Clarke and Kirkman, 1989). Shepherd et al. (1989) raised the possibility that the loss of 445 ha of *Z. tasmanica* in northern Adelaide waters between 1965 and 1985 was due to sediment accretion.

Seagrasses are also likely to be affected by the nature of the sediments that are deposited, which may bring with them pollutants or a high nutrient load. Furthermore, the redox state of sediments may be altered if there is a high organic load.

5. Seagrass recovery

Despite the known causes of widespread seagrass loss, few studies documented post-disturbance recovery rates of seagrasses (Campbell and McKenzie, 2004). The paucity of data on rates and extent of recovery of seagrass meadows is often due to the lack of data from long-term monitoring programmes and because many seagrass meadows have either failed to recover or taken many years to recover following stress from declining water quality (Short and Wyllie-Echeverria, 1996). In areas disturbed by dugong grazing, propellor scars and other small-scale disturbance, recovery can occur within weeks to months (Williams, 1988; Preen, 1995; Rasheed, 1999). Recovery of subtidal seagrass meadows from large-scale disturbance has been shown to take 2–4 years (Preen et al., 1995) or more than 5 years (Birch and Birch, 1984; Onuf, 2000; Blake and Ball, 2001; Frederiksen et al., 2004; Sheridan, 2004). Often, denuded areas may not recover for many decades because of chronic turbidity due to continual resuspension of unconsolidated sediments (Thorhaug and Austin, 1976). When water quality conditions do not return to their original state, recovery of subtidal seagrass may not occur at all (Giesen et al., 1990).

Campbell and McKenzie (2004) reported on the loss and subsequent recovery of approximately 2000 ha of intertidal *Zostera capricorni* beds from Great Sandy Strait in Queensland, Australia, following substantial flooding of the Mary River in 1999. Whereas 95% of the seagrass meadows in this region were lost within 6 months following the flood due to 2–3 fold increases in turbidity and nutrient concentrations, full recovery (through recolonization from seed banks) occurred within 2 years of initial loss (Campbell and McKenzie, 2004). Recovery of 2000 ha of primarily *Halodule wrightii* due to improved water quality conditions has been reported for Tampa Bay, Florida (Johansson and Lewis, 1992; Lewis et al., 1998). Approximately 1000 km^2 of seagrasses in Hervey Bay, Australia, were lost in 1992 after two major floods and a cyclone within a three-week period, which caused a persistent plume of turbid water. The deepwater seagrasses apparently died from lack of light from the floods. Heavy seas also uprooted seagrass in shallow waters. Subtidal seagrasses (below 5 m deep) started to recover within two years. Intertidal seagrasses only started to recover after four to five years and did not fully recover until December 1998 (Coles et al., 2003). Deterioration of water quality in the Gulf of Adelaide (Australia) from sewage effluents, sewage sludge and stormwater discharges, caused the loss of more than 4000 ha of seagrass between 1969 and 1996 (EPA, 1998).

No significant recovery has been observed to date and losses are continuing, despite some improvements in sludge outfalls.

Variation between different seagrass species in their ability to endure and recover from periods of reduced light is related to their differing morphological and physiological characteristics (Chesire et al., 2002). These characteristics represent different strategies for survival in the face of stress or disturbance. Smaller fast growing (short-lived) species such as *Halophila ovalis* or *Halodule wrightii* do not endure long once environmental conditions are beyond that to which they can adapt, but they tend to recolonize more quickly following an impact. Larger seagrass species such as *Thalassia* or *Posidonia* sp. tend to have greater stored reserves that can be mobilised to sustain the plant temporarily during periods of reduced light (below their minimum light requirements). These species tend to be slow growing, long-lived and therefore represent a resilience strategy, being more resistant to short-term to medium-term disturbances. If, however, the impact persists to the point where these plants have depleted all their reserves, they die. Once lost, recolonization of these species is unlikely or at best slow (Chesire et al., 2002).

6. Scale of damage to seagrass beds from dredging

An overview of 45 documented cases of dredging operations in or near/around seagrass areas, including scale of damage (ha) and mitigating measures applied (if any), is presented in Table 4. A total of 26 out of the 45 case studies presented, together account for a total loss of 21,023 ha of seagrass beds due to dredging and associated activities during the past 50 years. A further 12 case studies reported adverse (in some cases catastrophic) impacts from dredging operations on seagrasses, but did not quantify the total area lost. In the remaining seven case studies, no significant impacts of dredging on nearby seagrass beds were reported. Most of the reported case studies were in Australia (15) and USA (14), with the rest (16) scattered over Europe, Asia, Caribbean and the Middle East.

There must be many more cases of seagrass loss associated with dredging operations worldwide which are mostly – if at all – reported in gray literature and EIA reports, including confidential documents or reports in other languages, access to which is limited. The actual scale of dredging damage to seagrasses worldwide can therefore safely be assumed to be much greater. For example, recent large-scale dredging and land reclamation works in Singapore covering over 10,000 ha (De Jong et al., 2005) are likely to have caused damage to seagrass beds, but this has not been documented. Dredging to purposely remove "unwanted" seagrass vegetation is commonly practised in the Maldives at resorts without restriction (Iain Benson, pers. comm., May 2005). Seismic explorations for oil in Belize on turtle grass flats (several decades ago) resulted in permanent dotted lines across several of the grass flats, each dot the size of a 2-car garage (Anonymous, in lit.).

More recently, a series of some of the largest land reclamations in recent history have been initiated in Dubai, United Arab Emirates (De Jong et al., 2005). The dredging and dumping operations that are necessary for these extreme reclamations in Dubai (covering well over 20,000 ha), apparently permitted under governing national legislation, and several other recent reclamation and coastline modification projects elsewhere in the Middle East, are likely to affect large areas of sensitive marine habitats including seagrass beds (B. Riegl., pers. comm., May 2005; Purkis and Riegl, 2005).

Nevertheless, the selection of case studies presented here is considered representative of the scale and nature of damage commonly occurring. Other factors such as land reclamation, port construction, beach nourishment and poor catchment management also contributed to seagrass decline. Yet in all of these cases, dredging reportedly played a key role.

Two of the largest losses, i.e. 15,000 ha in Laguna Madre (Texas, USA) and 'thousands of hectares' in Moreton Bay (Australia), have occurred before 1990. Most of the more recent losses in Australia and USA are substantially smaller in scale and 6 of the 7 no-impact case studies were recent (post-1990). Data from European countries and elsewhere are too few to detect similar trends.

As the various case studies reveal, the extent of damage to seagrasses is not simply a function of the size and scale of the dredging operation alone, but also depends on the proximity to the seagrass bed, type and composition of the sediment, the way dredging equipment is used, mitigating measures applied, and so forth.

The scale of predicted damage to seagrass vegetation and cumulative effects can be issues of concern in the permit process for dredging and sand mining. According the Florida Fish and Wildlife Conservation Commission, for example, over 200 permit requests are submitted each year for small-scale dredging and constructions in Florida that may affect very small areas (often below 100 m^2) of seagrass vegetation (FFWCC, 2001; Kirsch et al., 2005). No amount of seagrass loss, no matter how small, is allowed to happen in Florida without formal permits being issued.

In several cases in the USA, applicants have attempted to seek dredging permits to fit channels into gaps in seagrass beds. Yet, opponents have argued (successfully in some cases) that seagrass meadows are known to migrate across the landscape and dredging below the compensation depth in a gap would eliminate potential seagrass habitat in the future (Jud Kenworthy, pers. comm., April 2005). Concern over predicted impacts on areas that constitute potential seagrass habitat but are currently unvegetated is sometimes considered legitimate in areas where major efforts are underway to restore seagrass vegetation, as noted for Tampa Bay (FFWCC, 2001) and the Dutch Wadden Sea (Van Katwijk, pers. comm., December 2004).

Not all observed reductions in seagrass cover in the immediate vicinity of dredging sites are necessarily the

Table 4
Overview of case studies on dredging impacts on seagrasses

Country	Location	Year	Activity/Purpose	Scale of impact/damage	Mitigation/Response	Reference
Australia	Botany Bay, New South Wales	1942–1984	Widespread dredging, along with poor catchment management and uncontrolled effluent disposal	Loss of **257 ha** of seagrass beds (*Posidonia australis*, *P.sinuosa*)	None reported	Walker and McComb (1992), Larkum and West (1990)
Australia	Success Bank and Parmelia Bank, Western Australia	1950s–2002	Commercial dredging for mining of calcium carbonate sands (shellsand) for production of lime (for mining industry) by Cockbum Cement	Loss of **232 ha** of seagrass (*Posidonia coriacea, Amphibolis griffithii* and *Posidonia sinuosa*) + additional loss of 168 ha predicted for period 2002–2014	Transplanting seagrass sods using planting machines for rehabilitation	Gordon et al. (1996), Wyllie et al. (1997), Lord et al. (1999), Paling et al. (2001), Walker et al. (2001)
Australia	Southern Bay Islands Region, Moreton Bay, Queensland	1955–2000	Dredging of access channels and marine infrastructure development (canal estate development)	Loss of thousands of hectares of seagrass (*Zostera capricorni*) due to dredging and associated turbidity	None reported	Kirkman (1978), WBM, 2001, Thorogood et al. (2001), Coles et al. (2003)
Australia	Cleveland Bay and Magnetic Island, Queensland	1970s	Capital and maintenance dredging at Ross River mouth and disposal at various dump sites in Cleveland Bay (peak in the early – mid-1970s)	Extensive burial and loss of nearly all seagrass vegetation (possibly several thousand ha according to habitat maps) followed by gradual recovery during 1978–1985	None reported	Pringle (1989)
Australia	Section Bank (Barker Inlet), South Australia	Late 1980s	Channel dredging of Port River	None reported (impact considered acceptable)	None reported (impact considered acceptable)	Chesire et al. (2002)
Australia	Section Bank (Barker Inlet), South Australia	Mid-1980s	Dredging of a trench for the Wasleys to Adelaide Pipeline Looping Project (30–50,000 m^3)	No significant (long-term) effects on seagrass	None reported	Chesire et al. (2002)
Australia	Deception Bay, Queensland	1991–1992	Channel deepening and maintenance dredging of the access channel into Newport Waters Canal Estate	No significant impacts detected	Monitoring (3yrs.) and mapping	Long et al. (1996)
Australia	McArthur River, Northern Territory, Western Gulf of Carpentaria	1994	Capital dredging of 1,250,000 m^3 of sediment for development of trans-shipment facility (incl. large channel, swing basin for berthing)	Loss of **18.95 ha** dense seagrass (direct removal) 3 years of monitoring indicated no further loss along canal edges or adjacent bed	Confined land disposal with 2-stage sedimentation ponds; monitoring of seagrass along channel edges	Kenyon et al. (1999)
Australia	Botany Bay, New South Wales	1994–1995	Dredging and landfill for construction of Sydney airport 3rd runway extension	Loss of **18 ha** of seagrass (*Zostera capricorni*) due to direct removal + additional loss of **5 ha** (*Posidonia* spp.) due to sand relocation for bird habitat reconstruction works	1.8 ha of seagrass transplanted successfully for compensation (pilot trial project)	Lord et al. (1999)
Australia	Port of Karumba, Queensland	1994–2004	Maintenance dredging of port entrance and river channel	No observable impacts on approx. 1000 ha of seagrass within the Port area	Long-term monitoring	Rasheed et al. (2001)
Australia	Fisherman's Island, Brisbane, Queensland	2000	Dredging and filling for the proposed expansion of the Port of Brisbane (dredging of 300,000 m^3)	Direct loss of **90 ha** of seagrass (patchy) predicted	Environmental management plan	POB (2000), BREC (2000)
Australia	Port of Weipa, Queensland	2000–2003	Capital and maintenance dredging of 3,750,000 m^3 to widen and deepen entrance channel and berth facility	Minimal or no impact predicted on 4000 ha of seagrass within Port limits	If visual plume over seagrass persists >6 h dredging to be relocated (+plume modelling)	GHD (2005)

Australia	Owen Anchorage, Cockburn Sound Western Australia	2002–2010	Planned Stage One-Dual Channel Dredging by Cockburn Cement (commercial shellsand mining)	Estimated direct loss of **53 ha** of seagrass (of which 38 ha *Posidonia sinuosa* and *Posidonia australis* +15 ha *Posidonia coriacea* and *Amphibolis griffithii*)	Detailed environmental management plan and transplanting proposed	Lord & Associates (2000)
Australia	Towra Beach, Botany Bay, Sydney	2004–2005	Dredging and filling for the Towra beach nourishment project (60,000 m^3 using cutter suction dredge) + dredging for the parallel runway project	Predicted loss of **3.85 ha** of seagrass (*Zostera capricorni*) due to the nourishment project + loss of **13.73 ha** due to the parallel runway project	Minimize overall seagrass loss and avoiding all *Posidonia* seagrass beds	SMEC (2003)
Australia	Port Philip Bay, Melbourne	2005	Dredging for channel deepening, Port of Melbourne (31.7 million m^3 sediment plus 0.5 million m^3 rock)	No significant impacts on seagrasses expected (20% reduction of primary production acceptable)	Seagrass productivity not to be reduced by more than 20%; turbidity to be kept within thresholds; extensive monitoring program	Edmunds et al. (2004), Pert of Melbourne Corp., 2004, Hart et al. (2004)
Bahrain	Fasht Al-Adhm, east coast of Bahrain (Arabian Gulf)	1985–1992	Dredging and filling associated with various land reclamations along the north-east coast	Loss of 10.2 km^2 (**1002 ha**) of seagrass beds detected from remote sensing imagery	None reported	Zainal et al. (1993)
Bermuda	Castle Harbour, Bermuda	1942–1943	Dredging and fill operation (12–15 million m^3) for a 300 ha land-fill for army station and Bermuda International Airport	Loss of **18.2 ha** of seagrass (*Thalassia testudinum*) due to dredging and associated turbidity	None	Smith (1999), Sterrer and Wingate, 1981
Brazil	Sepetiba Bay, Rio de Janeiro State	1997	Dredging of 20.86 million m^3 of bottom sediment to increase the capacity of the Port of Sepetiba	Accumulation of heavy metals by seagrasses from resuspended contaminated sediments	None reported	Filho et al. (2004)
Denmark	Saltholm and surrounding waters (Øresund)	1995–2000	Dredging and reclamation for the construction of the Øresund fixed link (bridge and tunnel) between Denmark and Sweden	No impacts on eelgrass beds (*Zostera marina*) (zero loss)	Feedback monitoring (stopping the dredging when turbidity thresholds exceeded); strict environmental regulations and extensive monitoring	Thorkilsen and Dynesen (2001), Jensen and Lyngby (1999), Krause-Jensen et al. (2001)
Fiji	Suva region, Fiji Islands	Early 1980s	Commercial dredging of coral sand for cement production (100,000 tonnes dry weight/yr in 1981)	Total destruction of seagrass beds within the dredgepit areas (not quantified) but gradual recolonization of dredged areas over time	Management plan	Penn (1981)
France	Gulf of Porto-Vecchio, Corsica	1970s	Dredging and port construction for Porto-Vecchio's commercial port	Almost complete disappearance of *Posidonia oceanica* beds in far end of Gulf of Porto-Vecchio	None reported	Pasqualini et al. (1999)
Hong Kong (China)	Chek Lap Kok International Airport, Hongkong	1994–1998	Dredging, reclamation and construction works for new international airport	Complete disappearance of seagrass (*Zostera japonica*) but some local recovery of *Halophila ovata*	None reported	Lee (1997), Fong (2000), Fong (2001)
Indonesia	Benoa Bay, Bali	1996–1998	Dredging (50 million m^3) and filling for land reclamation (Bali Turtle Island Development and Bali Benoa Marina)	Substantial loss of seagrass beds (approx. **500 ha**-estimated from habitats and development map)	EIA; Project abandoned after completion of reclamation in 1998 due to financial crisis	Shaw (2000)
Italy	Gulf of Oristano, Sardinia	1970s	Channel dredging and commercial port construction	Substantial loss of *Posidonia oceanica* meadows (approx. **800 ha** estimated from distribution map)	None reported	De Falco et al. (2000)

(*continued on next page*)

Table 4 (*continued*)

Country	Location	Year	Activity/Purpose	Scale of impact/damage	Mitigation/Response	Reference
Italy	Cape Feto, SW Sicily (Mediterranean)	1993	Dredging and fill operation for construction of Italo-Algerian methane gas pipeline	Direct loss of **150 ha** of seagrass (*Posidonia oceanica*) plus indirect effects from pulsed siltation on nearby seagrass meadows	None reported	Di Carlo et al. (2005), Di Carlo et al. (2004), Badalamenti et al. (2006)
Italy	Ischia Island, Gulf of Naples (Tyrrhenian Sea)	2002	Sand extraction for refilling of a beach	Loss of **4 ha** of *Posidonia oceanica* off Ischia Island (direct loss)	None reported	Gambi et al. (2005)
Kenya	Mombasa coastal area	Mid-1990s	Dredging and filling associated with jetty construction by local fishing company	Loss of **2 ha** of seagrass (5 spp.) plus additional (indirect) impacts from associated turbidity	None reported	Wakibya (1995)
Netherlands	Hond-Paap tidal flat, Ems estuary	2002–2003	Dredging and excavation of 250,000 m^3 of sediment for the deepening of an existing gas pipeline	No significant impacts on nearby eelgrass beds (*Zostera marina*)	Restrictions to timing, turbidity plume modelling, EIA monitoring programme	Erftemeijer (2002), Erftemeijer and Wijsman (2003)
Portugal	Rio de Aveiro, Atlantic Ocean	1984–2003	Channel dredging (deepening and widening) of the inlet connecting the Rio de Aveiro estuary/lagoon with the Atlantic Ocean	Loss of 8 km^2 (**800 ha**) of seagrass vegetation (*Potamogeton pectinatus, Ruppia cirrhosa* and *Zostera noltii*) due to indirect effect of dredging on turbidity, resuspension and tidal wave penetration	None reported	Da Silva et al. (2004)
Portugal	Rio Formosa (tidal/coastal lagoon)	Mid-1990s	Dredging of navigation channels and two tidal inlets to facilitate ocean passage and increase tidal circulation	Some seagrass affected (not quantified) (*Cymodocea nodosa* and *Zostera marina*)	Short-term monitoring	Janelle Curtis (in litt)
USA	Boca Ciega and Tampa Bay, Florida	1876–1976	Channel deepening, maintenance dredging, shell dredging and dredging for landfill and construction	Loss of 1400 acres (**567 ha**) of seagrass areas (5 spp.); extremely slow recovery (partly due to loss of offshore sandbars)	Complete halt of open water spoiling in Tampa Bay since 1973; use of upland disposal sites (at increased costs)	Lewis (1976), Taylor and Saloman (1968), Fonseca (2002), Lewis et al. (1998)
USA	Indian River Lagoon, Florida	1940–1992	Maintenance dredging and creation of spoil islands	Burial of seagrasses by creation of spoil islands and further losses due to turbidity from dredging (but some recolonization by *Halodule wrightii*)	None reported	Fletcher and Fletcher (1995), Brown-Peterson et al. (1993)
USA	Laguna Madre, Texas	1965–1988	Maintenance dredging (every 2–5yrs) of navigation channels and disposal of spoil (elsewhere in lagoon)	Loss of **15,000 ha** of seagrass beds (*Halodule wrightii*) from all waters deeper than 1 m due to turbidity caused by dredging/disposal with very little recovery	Various management actions and research on seagrass recovery	Onuf (1994), Quammen and Onuf (1993), Pulich and White (1991)
USA	Port of Miami, Florida	Early 1980s	Channel deepening, dredging of turning basin and filling of an artificial island as part of the expansion of the Port of Miami	Loss of **33 ha** of seagrass (+ additional loss of 69 ha of potential seagrass habitat)	The permit required replanting of 102 ha of seagrass to mitigate for losses; initial success very poor (2.4 ha survived)	Gaby et al. (1986), Lewis (1987)
USA	Key Biscayne, Florida	1985	Dredging and placement for beach nourishment of 3.9 km of Atlantic beaches	Burial and loss of **10.5 ha** of seagrasses	Salvage of seagrasses for use in mitigation of another dredging project (Port of Miami)	Gaby et al. (1986)
USA	Great Bay estuary, New Hampshire	<1993	Dredging and construction for expansion of the New Hampshire Port	Loss of **2.5 ha** of seagrass (Zostera *marina*) from direct and indirect impacts of dredging	Restoration of 2.5 ha of seagrass bed elsewhere in the estuary + long-term monitoring	Short et al. (2000), Davis and Short (1997)

USA	Laguna Madre, Texas	1994–1995	Maintenance dredging and disposal of dredged material (total 715.500 m^3) at six sites in the lagoon	Burial of seagrass vegetation at disposal sites; significant recovery (*Halodule wrightii*) within 3 yrs	Monitoring of recovery rate	Sheridan (2004)
USA	Indian River Lagoon, Florida	1996–1997	Fort Pierce cargo port extension, incl. deepening of the harbour and deepening and widening of the entrance channel	Loss of 39.3 acres (**16 ha**) of seagrass (*Halophila johnsonii*) + indirect secondary impacts from turbidity on highly productive seagrass beds (400 acres) nearby	None reported	MacArthur Report (1997), Virnstein and Morris (1996)
USA	Delmarva Peninsula, Maryland and Virginia	1996–1999	Hydraulic dredging and modified oyster dredging for clams (fishing)	**1257 ha** of seagrass affected/ damaged with scars; slow recovery taking >3 years	Adoption of legislation for the protection of most seagrass beds in Virginia (1997) and Maryland (2002) not allowing clam dredging in seagrass beds	Orth et al. (2002)
USA	Tampa Bay, Florida	1999–2000	Dredging for navigation and berth improvements as part of Port Manatee expansion project	Loss of 5.33 acres (**2.2 ha**) of seagrass (*Thalassia testudinum*. *Halodule wrightii* and *Syringodium isoetifolium*)	Transplanting of 17.57 acres (=7.1 ha) of seagrass (mainly *Halodule wrightii*) achieved within 3 years	Environmental Affairs Consultants (2005)
USA	Cape Ann peninsula, Massachusetts, Manchester-on-the-Sea	2001	Channel maintenance dredging to improve harbor access	Initial loss of seagrass in dredged channel areas; good post-dredging recovery within 3–4 years	Minimizing duration; seasonal restrictions; no-spud zone; limit over-dredge quantities; 5-year seagrass monitoring program	Peña (2005)
USA	Emeryville Flats, San Francisco Bay	2001–2002	Dredging, filling and construction related to the San Francisco – Oakland Bay Bridge project	58% decline in vegetation cover representing a loss of **8.3 ha** of seagrass (*Zostera marina*)	Not available	Merkel (2003)
USA	Miami Harbor area, Biscayne Bay	2002–2003	Dredging for widening of entrance channel and turning basin at Miami Harbor	Loss of 6.3 acres (2.6 ha) of seagrass by direct removal and subsequent sloughing	Replanting of 6.3 acres of seagrass (*Halodule wrightii* + 3 other spp.) on former borrow areas in Biscayne Bay	Dial Cordy and Assoc., 2002
USA	Scituate Harbor, MA, New England	2002–2003	Navigation maintenance dredging of small boat harbor	Loss of 1.8 ha of eelgrass (*Zostera marina*); some subsequent recovery within 2 years	No dredging allowed near dense eelgrass; use of silt curtains; seagrass monitoring	Sabol et al. (2005), Sabol and Shafer, 2005
US Virgin Islands	Water Bay, St. Thomas, Virgin Islands	1969	Dredging for navigation and boating (removal of 600,000 cubic yards of material from Water Bay)	Mass mortality/loss of seagrasses (*Thalassia testudinum*) and corals due to meachanical removal, sedimentation and turbidity	None reported	Van Eepoel (1969)

result of dredging-induced turbidity. Indeed, distinguishing effects of anthropogenic disturbances from natural dynamics in estuarine and marine environments can be a challenge (Montagna et al., 1998). Recent field monitoring at two sites in New England and Florida indicated that dredging-induced turbidity did not extend to nearby seagrass beds, and that locally observed seagrass decline must have been due to some other (natural) disturbance (Sabol and Shafer, 2005).

7. Mitigating measures

A number of management techniques and mitigation measures have been developed, such as tidal dredging, physical barriers, environmental dredging techniques and so forth, which may be used to mitigate effects of dredging on sensitive organisms or ecosystems (Smits, 1998). In hydraulic dredging techniques, the dredging rate can be adapted by increasing the amount of water pumped up relative to the amount of sediment that is dredged, which can help to reduce the extent of turbidity plumes. Examples of other environmental dredging equipment include encapsulated bucket lines for bucket chain dredgers, closed clamshells for grab dredgers, auger dredgers, disc cutters, scoop dredgers and sweep dredgers (all modified cutter dredgers). A more recent development is sub-suction dredging (e.g. BeauDredge, Multilans), which allows for lowering of the seafloor by extracting sediment from deeper layers without disturbing the top layer.

Mitigating measures applied in the various case studies (Table 4) include confined land-disposal, EIA, turbidity modelling (plume prediction), turbidity thresholds, limits to allowable reduction in seagrass productivity, minimizing duration of dredging, seasonal restrictions (e.g. avoiding seagrass flowering periods), limiting over-dredge quantities, establishment of no-spud zones, use of silt screens, prohibiting dredging near dense seagrass areas, stopping dredging when turbidity thresholds are exceeded, seagrass monitoring and mapping, research on seagrass recovery, salvage of seagrasses for use in transplantation to mitigate losses (Lewis, 1987), post-dredging seagrass restoration (Lewis, 1987; Fonseca et al., 2002), and adoption of legislation banning the use of certain (clam) dredging methods.

In the case of the Øresund Fixed Link (Denmark), a whole range of technical and environmental aspects of the dredging operation were integrated with the contractual commitments of the contractor to prevent impacts on eelgrass beds in the area (Jensen and Lyngby, 1999). Two major tools were introduced to ensure that spill was kept below the limits necessary to fulfill the environmental objectives and criteria of the project: (1) the contractor was held responsible through his contract for keeping the spill below specified limits varying in time and space, taking into consideration environmentally sensitive periods and areas; (2) a feedback-monitoring programme was implemented to covering sediment spill, dispersal thereof, and biological key variables representing the most sensitive benthic communities. Dredging was stopped temporarily during peak currents for approximately 20 times to keep within these environmental restrictions (Thorkilsen and Dynesen, 2001). These measures helped to ensure that there were no significant impacts from dredging and construction activities on the eelgrass beds (Krause-Jensen et al., 2001).

In at least one case, silt curtains have been used to reduce impacts on seagrasses (Sabol et al., 2005). Although the efficacy of silt curtains in reducing dredging-induced turbidity and siltation impacts on the seagrass beds was not evaluated, the authors note that the design of the curtains installed by the contractor may have been inadequate for the hydrodynamic conditions prevalent in the study area. High tidal flow resulted in breakage of anchor lines and rupture of the seams of the curtains (Sabol et al., 2005).

Installation of physical barriers such as silt screens is often a difficult operation, demanding great skill and experience on the part of the dredging contractor to avoid leakage through the curtain. Enclosure of dredging equipment with a silt screen is restricted mainly to the use with stationary dredgers using pipeline discharge methods, and is always accompanied by some degree of leakage underneath. Protection of an environmentally very sensitive area with silt screens may in some cases be viable, but only if the physical conditions of the site (esp. waves and currents) allow their effective use (USACE, 2005). Use of a silt screen, however, clearly limits the output level of the dredger, lengthens the execution period, and increases the costs of the project (Smits, 1998).

Impact prediction (e.g. through plume modelling and/or habitat modelling) and turbidity monitoring have proven to be successful tools in preventing or minimizing environmental impacts on seagrasses from dredging operations (Jensen and Lyngby, 1999; Thorkilsen and Dynesen, 2001; Krause-Jensen et al., 2001; Erftemeijer, 2002; Erftemeijer and Wijsman, 2004). A new method for synoptic real-time nowcasting, hindcasting and short-term forecasting of turbidity by combining information from remote sensing data, water quality modelling of sediment transport and in situ data using data-model integration (DMI) techniques, may prove to be an even more sophisticated, yet practical tool for use by the dredging industry in curbing environmental impacts (Erftemeijer et al., 2002; Tatman et al., 2005).

8. Regulation of dredging in seagrass areas

Specific regulations to protect seagrasses are few. Although marine protected areas (MPAs) are rarely established specifically to conserve seagrasses, there are 247 MPAs worldwide known to include seagrasses (Spalding et al., 2003). In addition to designation of MPAs, other legal measures have proved beneficial to seagrasses in some places, although seagrasses themselves are rarely singled out as the object of protection. In Queensland (Australia), for example, all seagrasses and other marine plants are specifically protected under the Fisheries Act of 1994, for protection of commercial and recreational fishing activities (Spalding et al., 2003). In the Mediterranean, *Posidonia oceanica* meadows are accorded priority protection under Annex 1 to the EU Habitats Directive, while 3 of 5 Mediterranean seagrass species are described as endangered in the SPA/BD Protocol of the 1995 Barcelona Convention (EEA, 2005). Since *Posidonia oceanica* meadows are particularly vulnerable to bottom-disturbing activities, some countries like Spain have severely limited dredging operations susceptible to alter *Posidonia* communities along the coast of Spain (Eurosion, 2004).

Both dredging and disposal operations are increasingly regulated more strictly with regards to their environmental impacts. In addition to national and regional legislation and policies, some useful general guidelines have been drawn up within the framework of international and regional conventions. The London Convention (1972) adopted the dredged material assessment framework (DMAF), a widely reviewed and accepted approach to the assessment of suitability of dredged material for disposal at sea. The OSPAR Convention (1992) adopted the OSPAR Guidelines for the Management of Dredged Material (OSPAR, 1998) and more recently produced a background document on 'Environmental Impacts to Marine Species and Habitats of Dredging for Navigational Purposes (OSPAR, 2004). Some other helpful documents include the 'Good Practice Guidelines for Ports and Harbours operating within or near UK European Marine Sites' (ABP Research, 1999), the 'Guidelines for Dredging' of EPA Victoria in Australia (EPA, 2001) and the series on 'Environmental Aspects of Dredging' issued by the Central Dredging Association (CEDA) and International Association of Dredging Companies (IADC) (Jensen and Mogensen, 2000).

In the United States, there are multiple levels of regulation of both direct and indirect effects of dredging and water quality degradation on seagrasses. At national level, the US Army Corps of Engineers has primary jurisdiction of dredging in "navigable waters", which includes essentially all marine and estuarine waters in the USA. Seagrass beds are regulated as "special aquatic sites" not as wetlands, but the same laws that apply to wetlands under Section 404 of the so called Clean Water Act (Federal Water Pollution Control Act, Public Law 92–500), originally passed by the US House and Senate in 1972, and modified since, apply regarding application for permits to impact any site with seagrasses. Permit applications are processed by the Corps, but are reviewed by other federal agencies, including the US Fish and Wildlife Service, National Marine Fisheries Service and US Environmental Protection Agency (EPA). EPA also has veto authority on any attempt by the Corps to override its recommendations for permit denial, although the power has been rarely used. EPA also regulates water quality directly under the same Clean Water Act with authority to issue permits to dischargers and limit quantities of pollutants, including suspended particles and nutrients, discharged to waters of the USA.

In addition, all of the 51 states of the USA, have individually some type of environmental pollution prevention laws, some stronger than others. In many states, like Florida and California, the Corps will not issue a federal 404 permit until the state issues a "water quality certification" that a proposed project, including dredging in or around a seagrass meadow, will not violate certain specific water quality criteria. Further some subdivisions of state government, such as counties or cities may have another set of regulations protecting wetlands in general and seagrasses specifically in coastal areas. All of these regulations require that permits be reviewed and if issued, include stringent monitoring of survival of seagrasses around dredging sites, and water quality monitoring during dredging. Turbidity, suspended particulate load, light penetration, dissolved nutrients and dissolved oxygen are common parameters that must be monitored and reported, and if exceedances occur, dredging must be stopped until monitoring confirms a return to background conditions.

In Australia, dredging and disposal of dredged spoil is governed by the Environment Protection Act 1981. In addition, the Great Barrier Reef Marine Park Act 1975 includes further provisions to assess and manage environmental impact of coastal development activities such as dredging. General dredging and sea dumping permits are subject to conditions which may typically include restrictions to the size of the impacted zone, limitations to a certain period of the year, turbidity limits and tidal restrictions (EPA, 2001). Turbidity limits are generally expressed in terms of maximum allowable exceedance (in %) above the best estimates of natural ambient turbidity. If this target is exceeded there are limits to the duration of plumes (e.g. not allowed to exceed an aggregate of four days in any five-day period, or 90% of any 10-day period). If such periods are exceeded, this will trigger a management response, such as temporary cessation or modification of dredging or disposal works or further restrictions on methods and location of future operations. Turbidity limits may be modified during the course of the dredging operation after consideration of monitoring results. Conditions also include the need for trial dredging, compliance- and effects monitoring (esp. of turbidity plumes and background turbidity), and monitoring and mapping of species composition and abundance of seagrass beds in the affected area. In most cases, a technical advisory and consulative committee is established that sets up and agrees on monitoring programmes, decides on the need and frequency of sampling and is allowed to make additional recommendations that will help to minimize environmental impacts of the dredging operations (EPA, 2001). Regulations may also include the need to rehabilitate affected seagrass areas.

Establishing a reasonable background turbidity level can be challenging, particularly given the variable nature of the ambient state. With a highly time variable background condition, some ports have adopted an approach of real-time comparison between plume measurements over sensitive areas with ambient conditions, whilst others continue to use an historic estimate based on regular monitoring of ambient conditions. Whilst dredging may cause an above average elevation of turbidity over seagrass, these elevations may in many instances be within the long term background range for the area and short-lived when compared to a frequent and naturally occurring event such as a significant rain event during the wet season. It can therefore not be assumed that transitory dredge plumes in excess of an area's average turbidity would necessarily cause significant impacts on seagrasses. Therefore, authorities increasingly take natural variability in background turbidity into consideration.

Historically, the management approach in the USA (with strong 'riparian' property rights anchored in constitutional laws) focused primarily on remediation (compensating for damage to be incurred to seagrass) with regulations applying to all dredging operations, even very small ones. In Australia, the approach has been more or less that of a zero-loss strategy, preventing or minimizing impacts on seagrass, with planned dredging operations being assessed following a case-by-case approach. Since about 15 years ago, the situation in the USA has changed, partly as a result of a growing awareness of economic losses associated with damage to seagrass beds. Now dredging in or near seagrass beds is very strictly controlled, and permits are hard to get, even for minor impacts (0.1 ha). Part of the issue is whether you can adequately mitigate or compensate for seagrass impacts. Most such projects have failed in the past (Lewis, 1987).

There is suspicion of potential bias and lack of objectivity in the licensing process for dredging permits. A recent study into sustainability of UK offshore dredging for marine aggregate mining (Olsen, 2005) noted that the vested economic interests of applicants, license granting bodies, license owners and monitors of the dredging are likely to affect objectivity in impact assessments submitted in the licensing procedure. Environmental impact assessment studies of large-scale land reclamation schemes in parts of the Middle East are sometimes shrouded in secrecy or are in some cases conducted only after reclamation has already been implemented.

9. Estimated costs

Stricter regulations, tighter control measures, proper enforcement and detailed monitoring, together with proper impact assessment, application of wide-ranging mitigating measures and the development of new environmentally friendlier dredging techniques appear to increasingly help in efforts to prevent or minimize adverse environmental impacts on seagrasses. Estimates of the extra costs of such mitigating measures (due to longer duration and special equipment needs) and/or compensation payments for damage incurred by dredging contractors are difficult to obtain. It appears reasonable, however, to assume that costs incurred in efforts aimed at preventing or minimizing seagrass damage are probably negligible in comparison with costs of seagrass restoration programmes, which often appear to have limited success.

Fonseca et al. (2002) discussed the ecological restoration of seagrasses including the success rate for seagrass mitigation undertaken for permitted losses of seagrasses, or as court ordered restoration after illegal damage. They noted that while seagrass restoration or mitigation can be successful, many errors occur in site selection in particular, and partial or complete failures in such attempts has been common. Citing a particular example of a court-ordered seagrass restoration project, they also note that successful efforts can be very expensive. The particular example they used costed US$ 630,000 per hectare (1996 costs) including site surveys, monitoring and reporting. Lewis et al. (2006) describe a US$ 6 million project that had only achieved approximately 1.5 ha of success at the time of reporting. A recent €900,000 eelgrass restoration programme in the Dutch Wadden Sea has achieved little success after 4 years of re-introduction and transplantation trials (Bos and Van Katwijk, 2005). The costs of two successful seagrass restoration projects in the USA ranged between 200,000 and 500,000 US$ per ha (Davis et al., 2002; Lewis et al., 2006). In their review of techniques for restoration and substitution of ecosystem services of tropical coastal ecosystems (including seagrass beds), Moberg and Rönnbäck (2003) conclude that it is probably always cheaper to aim at preserving ecosystem functioning than trying to restore or substitute them when they have been degraded or lost.

10. Discussion and conclusions

Although there clearly are a large number of reports that have documented adverse impacts on seagrass beds from dredging and sand-mining operations, there are several other (mostly recent) cases that reported no impacts on nearby seagrasses at all. There appears to be an increasing awareness among dredging contractors and regulatory bodies on the economic and ecological value of seagrass beds and the importance to make a concerted effort to minimize impacts on these sensitive systems. The various case studies from the USA and Australia suggests that the largest impacts have mostly been in the past. As a result of stricter regulations and enforcement by relevant authorities, recent large-scale dredging operations take various precautions and mitigating measures to keep impacts at a minimum. Elsewhere, experiences seem to vary between countries and from case to case.

As some of the case studies have shown, even large-scale dredging operations do not always cause significant impacts to seagrass beds. This may not necessarily be the result of strict environmental regulations and mitigating measures, but is sometimes a direct function of local environmental conditions.

Development of criteria to protect seagrasses must acknowledge that seagrasses tolerate periods of naturally high turbidity and can withstand some increase in the frequency of turbid events. Turbidity is unlikely to be continuous at any particular site particularly due to changes in wind and tidal conditions but also due to changes in dredge location and dredging rate. In areas that experience large natural fluctuations in background turbidity (esp. in estuarine environments), seagrasses and other benthic communities often display a greater resilience than in areas where natural turbidity fluctuations are minimal.

Nearshore seagrass communities, like other marine ecosystems, are commonly thought to be largely static entities, with predictable, seasonal changes. Increasingly, however, studies examining interannual variability in seagrass and other macrophyte communities remind us that this is not necessarily the case. Seagrass patches may come and go, may change position or density, and their associated fish communities may be different from year to year (Hemminga and Duarte, 2000). Consequently, what we observe in one year may or may not hold true in subsequent years. Given this potentially high variability, it is especially important to use caution when basing long-term management decisions on short-term observations. Ecosystems may not function or behave as expected when we base our expectations on a mere snapshot of a constantly changing entity (Weitkamp, 1998).

Therefore, copying of dredging criteria used in one geographic area to a dredging operation in another may in some cases lead to exaggerated limitations resulting in unnecessary costs and delays in dredging operations, or in other cases could prove damaging to seagrass ecosystems. Meaningful criteria to limit the extent and turbidity of dredging plumes and their effects will always require site-specific evaluations and should take into account the natural variability of local background turbidity.

Acknowledgement

The authors wish to acknowledge the following people who kindfully shared insights, practical experience, literature and information for this review: Bill Allison, Iain Benson, Vicente Tasso Bermell, Sian Boyd, Nick Bray, Archie Carr, Guiseppe Di Carlo, Eugene Dunaevsky, Adam Gelber, Melanie Gomes, Jorrit de Groot, Oliver Gussman, Emma Jackson, Maruf Kasim, Marieke van Katwijk, Donna Kendall, Jud Kenworthy, Hugh Kirkman, Barry Kwok, Eva Olsen, Bob Orth, Eric Paling, Johan Pennekamp, Allison Perry, Joanna Pitt, Gerard van Raalte, Chris Ryan, Marcello Sano, T. Shearer, Richard Snyder, Mark Spalding, Jason Tanner, Anitra Thorhaug, Simona Trimarchi, Nicky Villars, Robert Virnstein, Di Walker and Simon Wilson. This review was carried out as part of the research theme 'large-scale dredging operations' (cluster "optimalisation dredging technology in relation to prevailing turbidity criteria and soil type"). We kindfully acknowledge the financial support provided for this research programme by the Netherlands Ministry of Economic Affairs within the framework of the WL | Delft Hydraulics R&D Programme (EZ doelsubsidie 2005).

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Detection of decadal shoreline changes along dhamra and maipura coast, Odisha, India

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Proceedings of the 19th IAHR-APD Congress 2014, Hanoi, Vietnam

ISBN 978604821338-1

DETECTION OF DECADAL SHORELINE CHANGES ALONG DHAMARA AND MAIPURA COAST, ODISHA, INDIA

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ABSTRACT

Coastal erosion is one of the major problems of the coastal zone. The erosion is triggered by various reasons such as high wave energy, reduction of sediments, natural disasters and climate change etc. In the era of industrialization, major infrastructure developments are happening along the coast. Prior to the initiation of those projects, it is important to understand the coastal processes of erosion, deposition, sediment-transport, flooding and sea-level-changes of the region which continuously alters the shoreline. These processes disturb the stability and productivity of aquatic environment and may have severe implications for proprietors. This study attempts morphological assessment of shoreline in the middle coastal plains of Odisha state on the east coast of India. Shorelines changes study is carried out at Dhamra and Maipura Coast to quantify erosion and accretion during years 1990 – 2012. Satellite derived remote sensing data of LANDSAT and IRS P6 for the years 1990, 2000 and 2012 were used in this study. Dhamra coast has been modified in between these years due to anthropogenic disturbance such as port development. Maipura coast which has large mangrove cover, Bhitarkanika national park carries its ecological importance. These coasts experienced erosion during 2000-2012 compared to 1990-2000. Accretion is noticed in the nearby river mouths. Temporal variation of sediments and frequent flood events will also be discussed in the paper. The detailed analysis reveals that the maximum erosion of 227 m, 47 m in a decade at Dhamra and Maipura coasts respectively. Area of Dhamra coast was under accretion during 1990-2000 and this area experienced erosion near Dhamra port just after the development of port in 2007. This study concludes that shoreline of study area is under high risk of erosion and inundation due to natural as well as anthropogenic activities in the area.

Keywords: Remote sensing; Erosion; Accretion; Odisha coast; DSAS.

1. INTRODUCTION

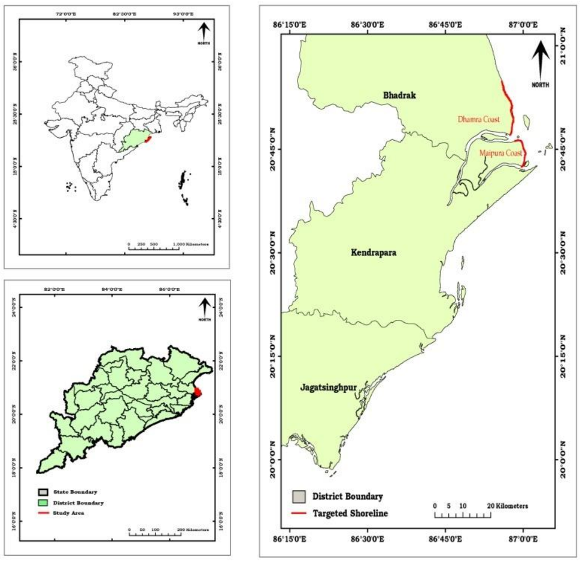
The coastline of India comprises of variety of habitats and ecosystems such as sandy and rocky beaches, cliffs, water and lagoons to bays, mangrove swamps, sea grass beds, coral reefs and estuaries. Indian coastline is about 7,500 kms in length and the EEZ is having an area of 2.02 million sq. km (Ramesh *et al.*, 2011). The recognition of a ''shoreline'' involves the selection of a shoreline indicator within the available data source (Ron *et al.*, 2001). Shoreline change is considered as one of the most dynamic processes in the coastal region and is caused due to various physical and anthropogenic processes (Chen *et al.*, 2005). Shorelines are always subjected to changes due to coastal processes, which are controlled by wave characteristics and the resultant near-shore circulation, sediment characteristics, beach form etc. (Kumar *et al.*, 2010). Assessment of long term erosion and accretion rate of the coastal area is essential for the selection of different types of coastal structures. Erosion and accretion index is prepared for Kuwait coast (Neelamani and Uddin, 2013). This study is helpful for identifying better sites for coastal infrastructural activities in the study area. Erosion has been observed at the region around the ports of Visakhapatnam, Paradip, Ennore and Nagapattinam on the east coast of India while deposition has been observed south of these ports. These changes are attributed to construction of artificial barriers like breakwater, jetties, etc. (Nayak, 1992). The rising number of coastal disasters along the world's coastlines throws light on the need for better and more efficient methodologies for the assessment of coastal vulnerability. A study at east coast of India was carried out to analyze and illustrate the vulnerability linked with various coastal hazards and can be used effectively by coastal managers and decision-makers to devise better coastal zone management plans as well as to ensure efficient mitigation measures to lessen the losses during disasters (Murali *et. al.*, 2013).

A fruitful management of the coastal region requires alert consideration of all the components of shoreline movement, as it is a complex phenomenon resulting from both natural processes and anthropogenic effects (Camfield and Morang, 1996). Construction of seawalls has resulted in shifting of erosion sites from one place to another, whereas, breakwaters have been acting as barriers for littoral drift at Mangalore coast (Kumar and Jayappa, 2009). Accretion was predominant along the coast between Kanyakumari and Tuticorin during 1969-1999, but this area had undergone erosion from 1999 onwards. Morphologic and hydrodynamic changes arise continuously subsequent to December 2004 tsunami (Mujabar and Chandrasekar, 2011). A recent study demonstrated shoreline changes and morphology of spits for southern Karnataka area, India (1910-2005) utilizing satellite data and statistical techniques which can be incredibly practical in quantifying shoreline changes and spit morphology (Kumar *et al*, 2010). Accurate quantification of the dimensions of material lost using erosion risk mapping assists decision making for the implementation of protective measures. A research was carried out to establish soil loss rates due to erosion by water and wind in protected natural areas, to predict the environmental effects of different land uses (Martinez-Grana *et al.*, 2014). Another study discussed an alternative cost-effective methodology involving satellite remote sensing images and statistics during their study for shoreline change analysis and its application to prediction (Maiti and Bhattacharya, 2008). DSAS is used for studying Quantitative analysis of shoreline changes at the Mediterranean Coast in Turkey. LANDSAT satellite data for the years 1972 (MSS), 1987 (TM) and 2002 (ETM) was used after image processing and coastline was detected by self organizing data analysis technique classifications, edge detection and overlay technique. DSAS was used to calculate erosion and accretion rate at different time intervals. A combined use of cartographic data and statistical methods could be a trustworthy technique for shoreline related studies (Kuleli, 2010). Application of such data seems to be trustworthy in qualitative monitoring of shoreline changes, while it is the only available method for long term studies (Bagdanavičiūtė *et al.*, 2012). Dynamic geomorphology of Mahanadi delta and problems of coastal dynamics and shoreline changes after the construction of Paradip port was studied (Meijerink, 1983; Rao, 1989). The extent of coastal geomorphological changes induced by the grounded ship MV River Princess was analysed on the Candolim–Sinquerim coastline of Goa, India (Murali *et al.*, 2013). In this study, dynamics behind erosion is discussed based on the southwest and northeast monsoon wave patterns and alignment of the ship with respect to the shoreline. (Rupali, 2007) studied the spit stability adjacent to the Jatadharmohan creek based on hydrodynamic conditions of the creek and slope stability. A study was conducted to understand nearshore erosion, deposition, sediment budget and longshore transports off Paradip area (Ananth and Sundar, 1990; Sarma and Sundar, 1988). Erosion, that is observed north of Paradip and Ennore ports on the east coast of India, is due to construction of artificial breakwaters and jetties (Nayak *et al.*, 1992, 1997; Chauhan *et al.*, 1996). Coastal processes along the Indian coast with reference to the erosion and accretion was studied (Sanilkumar, 2006). Another study was carried out to monitor the shoreline environment of Paradip using remote sensing for the period 1973-2005. The years 2001, 2002 and 2003 exhibited loss in length of shoreline as well as area of the beach (Murali *et al.*, 2009). The Objective of this study is to monitor and quantify the erosion and accretion for annual to decadal scales at Dhamara and Maipura coasts of Odisha coast using remote sensing data and GIS.

2. STUDY AREA

The area under investigation is the coastline of Odisha state located on the east coast of India. The Odisha coastline is 480 km in length and consists of six coastal districts. Dhamara and Maipura coasts are targeted for shoreline mapping in order to quantify the erosion and accretion during the period 1990 – 2012 (22 years). The study area (Figure 1) is located between 20°43'17.26" N - 20°2'51.875" N, and 87°4'6.915" E - 86°26'0.235" E. The study area has a tropical climate and summer maximum temperature ranges between 35-40° C and the low temperatures are usually between 12-14°C. The average rainfall is measured to be 1482 mm and receives an average of 78% of rainfall between the months of June and September and the remaining 22% of the rainfall throughout the year. The source of the sand that feed the beaches, dunes, and barrier beaches comes primarily from the erosion of coastal landforms (Ramesh *et al.*, 2011). A unique feature of the adjacent Bay of Bengal (BoB) is occurrence of tropical cyclones during October-November and April-May (Sehgal *et al.*, 1991). Storm surges that are generated by the cyclones in the Bay of Bengal cause tremendous destruction along the east coast of India. A study was carried out for projected sea level rise estimation for regions surrounding Nagapattinam, Kochi and Paradip. According to this study, Paradip is known for the occurrence of storm surges resulting from the passage of cyclones (Unnikrishnan *et al.*, 2010). The cyclones that affected the Orissa coast between 1877 and 1987 show irregular tracks and they occurred in between the mouth of the Dhamra River and Paradip. Between 1891 and 1970, there were 1036 depressions in the Bay of Bengal and among them, 360 intensified into storms (Ramesh *et al.*, 2011). The Mahanadi River deltaic coast is micro-tidal with a mean tidal range of 1.29 m. The currents measured in the coastal waters of Odisha indicate that the flow is towards south with speeds varying from 14-29 cms^{-1}. An average annual total sediment load of 29.77 million tons are carried by the Mahanadi River at its delta head (Kumar *et al.*, 2010). Mudflats, spits, bars, beach ridges, creeks, estuaries, lagoons, flood plains, paleo-mudflats, coastal dunes and salt pans are observed along the Mahanadi delta of Orissa.

Figure 1. Location of study area



3. MATERIAL AND METHODOLOGY

3.1 Data used

The LANDSAT satellite images were downloaded from Global land- cover facility website and IRS-R2 satellite imagery was purchased from National Remote Sensing Centre, Hyderabad. The following Table shows the various images that were procured, their resolution and date of pass.

Date and Year	Path/ Row	Satellite ID	Sensor	Resolution (m)	Resolution (m) after resampling
28.11. 1990	140-46	Landsat 5	TM	30	23.5
23.10. 2000	139-46	Landsat 7	ETM+	30	23.5
24.02. 2012	107-58	IRS-R2	L3	23.5	23.5

Table 1. Satellite data information used for Study

3.2 Methodology

The satellite images of the study region were searched and accordingly, the above listed satellite images were used for this study. The important factors which need to be considered for finalizing the satellite images are cloud cover, similar tide conditions, similar season data, uniform projection factors, etc. In this study, the above factors were considered for the available images. The satellite data undergoes radiometric correction and data scaling to enable maximum visual interpretation. All images were geo-coded with UTM projection WGS 84 datum parameters. Geo-coded data are images that have been rectified to a particular map projection and pixel size, and usually radiometric corrections are already applied. About 30 Ground control points were used for this purpose.

One of the objectives was to understand and predict annual- to decadal-scale shoreline change along the study area using remote sensing and GIS. The study was conducted for different years i.e. 1990-2000, 2000-2012 depending upon the data availability.

First of all Image processing was done for different satellite images in ERDAS Imagine 9.1. The study area is a part of different scenes of satellite imagery. Mosaic image was prepared for different scenes and then subset of study area was created for further processing. All satellite imagery was geo-referenced and RMSE error was less than one only. Resampling of all satellite imagery was done in Erdas Imagine 9.1 to prepare same pixel size (23.5 m) for different datasets.

Preprocessed satellite imagery is used to digitize shorelines. Maps are line digitized so that shoreline can be marked. These marked boundaries are then overlaid for different years and difference can be brought out using ARC GIS 10.1. DSAS is used to compute rate-of-change statistics for a time series of shoreline vector data (Thieler *et al*. 2009). Transects were created on baseline at a distance of 50 m for all shorelines under study area.

4. RESULTS

Mapping of shoreline lengths for Dhamara and Maipura coasts were done at a temporal scale of 22 years. Detailed results of length of shoreline are given below.

Table 2. Decadal shoreline mapping results

Sr. No.	Coast Name	Length in 1990 (km)	Length in 2000 (km)	Length in 2012 (km)	Net in 2012	Results (km)
1.	Dhamara	16.94	18.22	18.83	Gain	1.89
2.	Maipura	10.71	10.01	10.24	Loss	0.47

These results show that shoreline behavior is highly dynamic in this area. This can be due to deltaic characteristics of rivers and cyclonic activities in the area. Dhamara coast showed a net increase in length of 1.89 km in 22 years. Maipura coast confirmed the net loss of 470 meters in length of coastline. Erosion and accretion measurement was done using DSAS by creating transects at a fixed distance spacing of 50 meters. High distance spacing of transects increases the uncertainty in results and very low distance spacing increase the huge amount of data values. Therefore the transect spacing was taken 50 meter in this study. These results are calculated for 1990-2000 and 2000-2012. Each coast under study area shows different behavior under different time scale.

Dhamara Coast

A maximum erosion of 227 m at transect id 42 and accretion of 537 m at transect no 136 is observed in 1990-2000 at Dhamara coast. Transect id 69 shows 269 m erosion and transect id 150 shows 513 m accretion in 2000-2012 (Figure 2).

Maipura Coast

A maximum erosion of 47 m at transect id 44 and accretion of 183 m at transect id 66 is observed in 1990-2000 at Maipura coast. Transect id 02 showed 93 m erosion and transect id 186 shows 177 m accretion in 2000-2012 (Figure 3).

Dhamara coast has undergone an average erosion of 5.05 m/year from 1990 to 2000, which is increased in next decadal interval i.e. 8.57 m/year during 2000 – 2012. Net erosion increase during 2000-2012 was 34% as compared to 1990 – 2000 (). Dhamara port is constructed in 2007 is also located in the area under erosion. Erosion at this site might be because of anthropogenic activities. Middle part of the coast is accreted and erosion is observed in northern and southern part. The area near Dhamara river mouth is accreted; this may be because of heavy sediment transport activities at river mouth. Northern part of coast is under high erosion.

Maipura coast experienced an average erosion of 1.41 m/year from 1990 to 2000, which is increased to 3.87 m/year in 2000 – 2012. Net erosion increase during 2000 – 2012 was 12% as compared to last decade. This coast has sediment transport activities at both ends. North part of the coast is affected by Dhamara river and southern part is affected by Maipura river. In northern part near mouth of dharma, maximum erosion is observed and the area near maipura mouth is maximum accreted during 2000 - 2012.

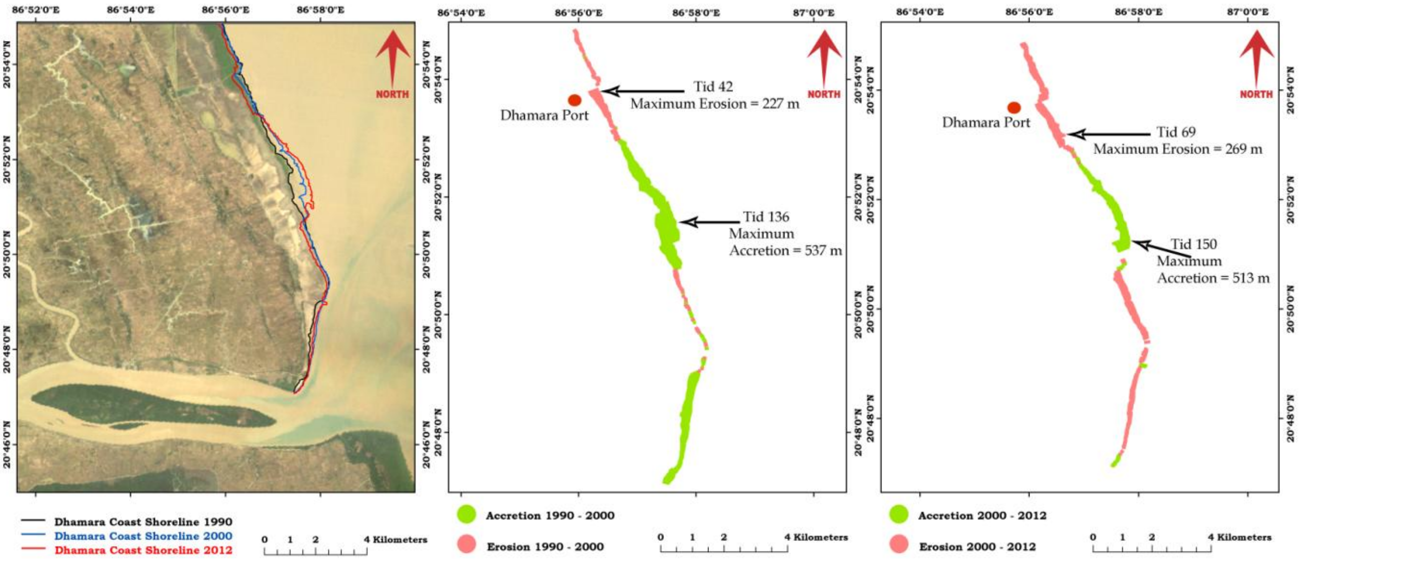
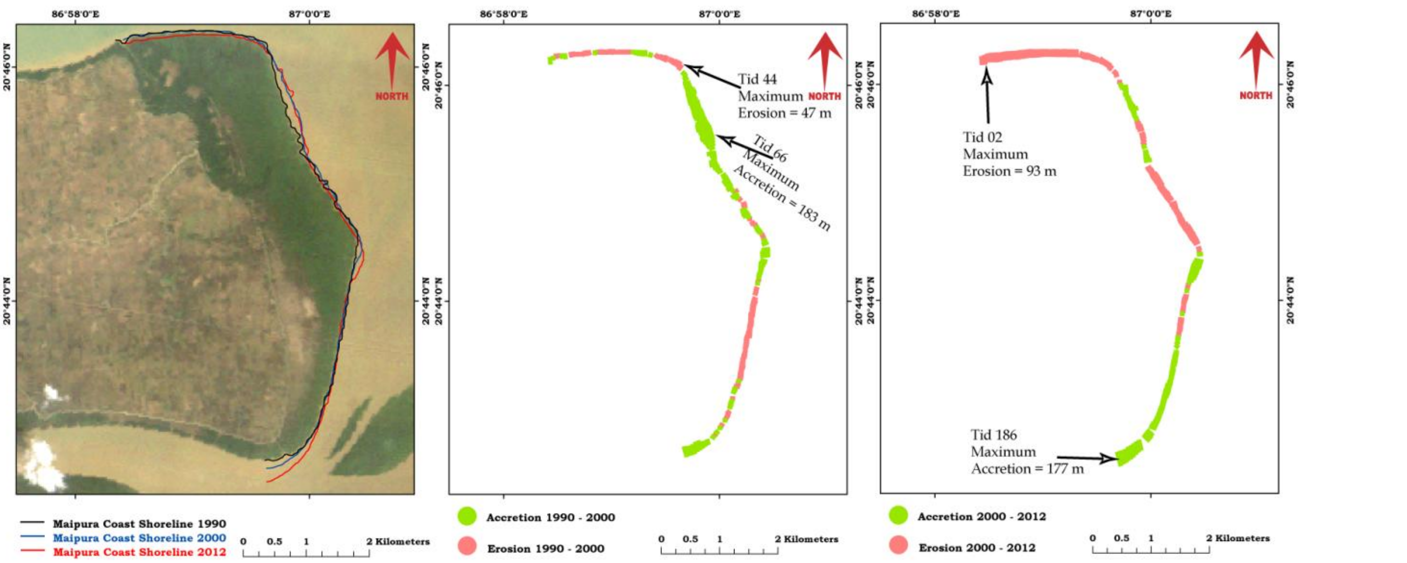


Fig 2. Erosion and accretion mapping for Dhamara coast for years 1990-2000 and 2000-2012



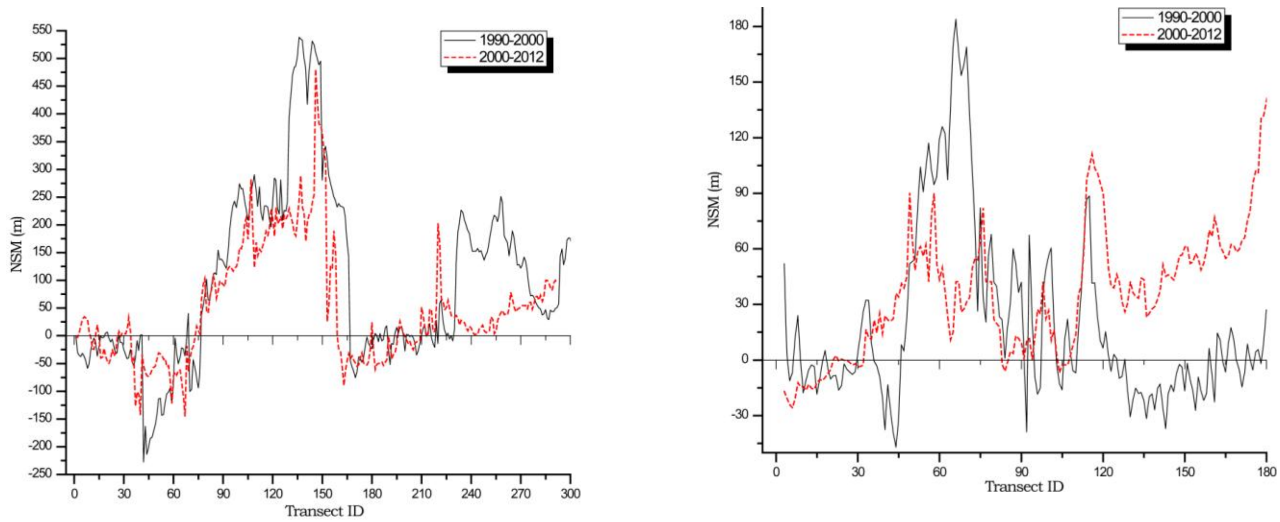


Fig 3. Erosion and accretion mapping for Maipura coast for years 1990-2000 and 2000-2012

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5. CONCLUSIONS

This study concludes that shoreline of Odisha coast is under high weight of erosion and this might be due to natural as well as anthropogenic activities in the area. Erosional activities increased during last few years compare to the last few decades. There may be some natural causes, for example – wave energy, cyclones from Bay of Bengal, floods in coastal plains and deltaic characteristics of rivers in the area. Area of Dhamara coast under accretion during 1990-2000 resulted into erosional area near Dhamara port just after the development of port in 2007. It is due to the anthropogenic involvements on the shoreline, the shoreline conservation measures are recommended near to the port areas. Dhamara coast is under accretion in northward direction and erosion is elucidating in southward direction, which shows dynamic ocean current flow direction from northward to southward. This result can be helpful in study of sediment transportation from Dhamara River into Bay of Bengal. Large mangrove area at Maipura coast revealed erosion, which is not a good indicator for biodiversity. Biodiversity conservation measures are recommended near the Maipura coast.

A physical approach of shoreline change detection is recommended for better accuracy of morphological assessment. Anthropogenic activities such as dredging, mining, etc. also contributed in quantum of erosion at study site. Studies of sediment dynamics from land into oceans by rivers in this region is also an inevitable in the study. Increased activities including anthropogenic emissions, erosions and their intended conservation draw the attention of scientific community to this region.

ACKNOWLEDGMENTS

Authors acknowledge Director, CSIR-National Institute of Oceanography, Goa, India, for the support of this study.

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8th International Conference on Asian and Pacific Coast (APAC 2015)

Impacts of Ports on shoreline change along Odisha coast

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Abstract

Ports involve construction of coastal structures such as groin, sea wall, breakwater, jetty etc. which result in modification of the shoreline and coastal geomorphology. Paradip and Dhamara are two major operational ports along Odisha coast while Gopalpur port is being developed as all weather open sea direct berthing port. The coastal structures associated with Paradip port at present are two breakwaters and northern sea wall while Dhamara port is built in an estuarine environment with one jetty. At Gopalpur Port, southern (530m) and northern (362m) groins were constructed during 2007-2009 along both sides of the previously existing piers. Other developments include intermediate (360m) and southern (1735m) breakwaters during November 2011 to December, 2012 on the south and groin field (ten groins of different dimension) on the north during September, 2011 to December, 2012. Impacts of these coastal structures are monitored every month from June, 2008 to April, 2014 by observing shoreline change, beach profile and Littoral Environment along 30km stretch on the north and south of Gopalpur port along with one year (June 2008-May 2009) wave and tide measurements by deploying tide gauge and wave rider buoy at 23m depth off Gopalpur port. Besides, seasonal observations on beach profile and shoreline near Paradip Port (August, 2010 to August, 2013) and long term (1975-2014) shoreline change study at Gopalpur, Paradip and Dhamara Port are carried out with topo-sheet data as base line information. The shoreline change rate (m/yr) in the pre and post construction phases are estimated which show distinct difference between the two phases and also on the north and south of the ports.

Peer- Review under responsibility of organizing committee , IIT Madras , and International Steering Committee of APAC 2015

Keywords: port, coastal structure, shoreline change, erosion/accretion

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Peer- Review under responsibility of organizing committee , IIT Madras , and International Steering Committee of APAC 2015
doi:10.1016/j.proeng.2015.08.339

1. Introduction

Coastal structures such as groin, sea wall, breakwater, jetty etc. result in modification of the shoreline and beach morphology (Elmoustapha et al., 2007) besides their significant impact on coastal dynamics (waves, currents etc.) and shoreline stabilization (Elsayed and Mahmoud, 2007). The effects of groin parameters on the shoreline change have been studied using both physical and numerical models (Ozolcer et al., 2006). Construction of hard structures along the coast either for development of ports and harbours, or for protecting the coast from erosion, significantly modifies the shoreline and the erosion/accretion trend. Groins are shore protection structures designed to trap longshore sediment for building a protective beach, retarding erosion of an existing beach, or preventing longshore drift from reaching some drowndrift point such as harbour or inlet. Groins modify the longshore sand transport and results in the accumulation of sand, mostly on the updrift side, and erosion of sand on the downdrift side. Mohanty et al. (2012) studied the impacts of groin on beach morphology at Gopalpur port (Fig. 1b) and revealed that the rate of deposition on the south of the groin is 2.5 times more than the rate of erosion on the north of the groin. Odisha Coast (Fig. 1a), with 57% sandy beaches, experiences erosion along 22.6% length of the coastline (Sanil Kumar et al. 2006). Erosion/deposition pattern of a shoreline critically depends on the annual gross sediment transport rate and direction. Contrary to previous studies, Mohanty et al. (2012) showed that the annual gross sediment transport rate varies between 1.10 x 10^6 m^3/year to 1.29 x 10^6 m^3/year and is unidirectional (towards north) round the year. Erosion at Paradip and north of it is attributed to the construction of breakwaters and groins at Paradip port (Fig. 1c) and closely similar to the erosion pattern in the northern regions of Chennai, Ennore and Visakhapatnam (Chauhan and Gujar, 1996; Nayak, 2004). Dhamara port (Fig. 1d) is built in an estuarine environment without breakwaters and groins and hence the impact of coastal structure on shoreline change is negligible. More detailed information on the geomorphological characteristics and shoreline change along Odisha coast can be found in Mohanty et al. (2008). Neither long-term nor short-term recent information on shoreline change and coastal erosion/deposition are available for Odisha coast, although Government of Odisha is contemplating to develop about 15 more ports. Therefore, information (past, present and future) on shoreline change and erosion/accretion are vital for successful design and management of ports, and also for coastal management. In the present study, our objective is to assess the impacts of coastal structures on shoreline change. Therefore, the shoreline change rate both in the pre and post construction phases of the three major ports along Odisha coast are studied. Besides, beach width and volume changes at seasonal and inter-annual time scales have been studied to quantify the erosion/deposition near the three major ports.

2. Survey and Computational Methods

Littoral Environmental Observation (LEO), beach profile and shoreline change were monitored every month from June 2008 to March 2011 and from June 2012 to March 2014 at Gopalpur port. At Paradip, observations were made during August, 2010, May, 2011, September, 2011, December, 2012 and August, 2013. At Dhamara, long term (1979-2014) shoreline change rate was assessed along with the seasonal (May and August, 2010) shoreline change at Ekakulanasi turtle nesting beach located south of Dhamara Port (Fig. 1d). Although beach profile were monitored every month over 24 transects on the south and 24 transects on the north of Gopalpur port, only seasonal (winter and monsoon) profiles at four representative transects on the south and four on the north are presented here for convenience. The distance between two consecutive transects were maintained at 500m. Leica SR 1200 Real Time Kinematic (RTK) Global Positioning System (GPS) was used for beach profile monitoring while berm position was monitored with the help of DGPS Arc Pad. Beach width and volume changes were computed using Beach Morphology Analysis Package (BMAP) of Coastal Engineering Design and Analysis System (CEDAS) software Version 4.0 developed by Veri-Tech Inc. The details of the instruments used and methods followed for beach profile and berm line measurements are same as in Mohanty et al. (2012). For shoreline change rate analysis, topo sheets of 1975, 1978 and 1979 have been used as the base line information for Paradip, Gopalpur and Dhamara respectively. Shore line change rate (ft/yr) at a later period were estimated using Regional Morphology Analysis Package (RMAP) of CEDAS with berm line information (observed) and/ or shoreline extracted from rectified Google image. Later shore line change rate computed in ft/yr (Figs. 3, 4b and 5) were converted to m/yr and is discussed in the text in order to maintain consistency in the use of units.

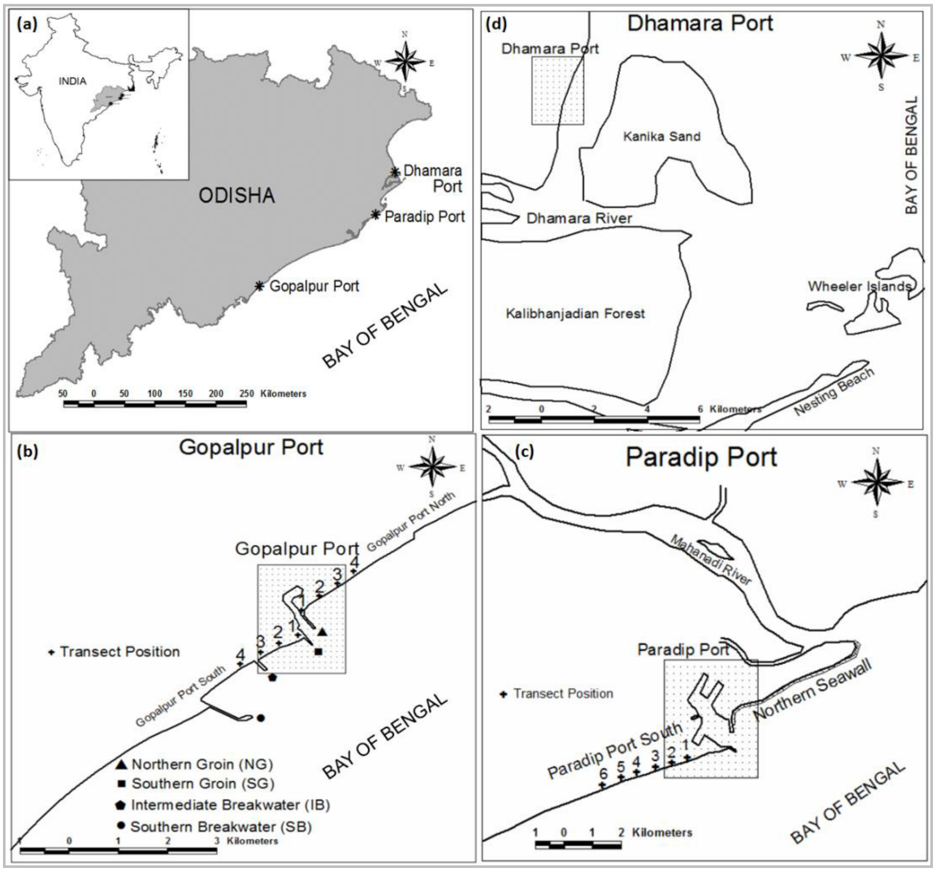

Fig. 1. (a) Map showing location of three major ports along Odisha coast; transect positions along (b) Gopalpur Port, (c) Paradip Port and (d) Dhamara Port.

3. Results and Discussions

3.1 Gopalpur Port

Gopalpur Port Limited (GPL) is located at latitude 19˚ 18′13″ N and longitude 84˚ 57′ 52″ E along Odisha coast, east coast of India (Fig. 1b). GPL is being developed as all weather open sea direct berthing port from a small fair weather port which existed since 1987. The structures associated with the fair weather port were a 400m northern pier and a 500m southern pier. Southern (530m) and northern (362m) groins were constructed on both sides of the previously existing piers during August, 2007 to November, 2009 and October, 2007 to September, 2008 respectively. Intermediate (360m) and southern (1735m) breakwaters were constructed respectively at a distance of 1.1km and 2.42 km south of southern groin (SG) during November 2011 to December, 2012. Northern groin field (ten groins of different dimension) extending up to 2.73km north of northern groin (NG) was constructed during September, 2011 to June, 2012. These coastal structures have definite impacts on shoreline which are discussed in the following sections.

3.1.1General Oceanographic conditions and LEO

The average spring and neap tidal ranges near Gopalpur Port are 2.39 m and 0.85 m respectively. The range is highest in August and lowest during January. Based on one year observations of waves, currents and tides near Gopalpur port, it is revealed that the average wave height ranges from 0.25 m in December to 0.97m in July.

The mean wave period varies from a minimum of 7.2 seconds in April–May to 10.8 seconds in December–January. The coast is exposed to waves approaching predominantly from the SSE, SE, and S. The significant wave height (Hs) varies from 0.26 m to 3.29 m with an average of 1.06 m during one year observation period. Hs varies from 0.4 m to 3.29 m with an average of 1.29 m during pre-monsoon and monsoon while the values are less (0.26 m to 2.18 m with a mean value of 0.71 m) during post monsoon and winter. Details of the oceanographic condition and LEO are also discussed in Mohanty et al. (2012).

3.1.2 Shoreline Change

Figure 2 depicts the shoreline change with time on the north and south of the port. Shoreline refers to the first berm during spring low tide while moving from foreshore to backshore which is not disturbed at the time of observation. Shoreline position (Y-axis) refers to the distance between the berm line and the common reference line drawn at the backshore nearer to vegetation line in a GIS environment. It is evident from Fig. 2 that the shoreline movement on the south of the port is seaward during the period of observation except during August, 2013 and January, 2014 corresponding to the position and period of berth development. Maximum seaward movement is observed immediately south of SG (0 km) which gradually reduces further south of SG. At port north the shoreline movement is landward except at 0km which is immediately north of NG. At 0km, oscillation of the shoreline is observed prior to the construction of northern groin field and southern breakwaters. However, the shoreline stabilised later due to beach fill by the dredged sand which continued after June 2012. Maximum landward movement of the shoreline is observed at 1km north of NG and further north the landward movement gradually decreased due to construction of northern groin field. After the construction of breakwaters and groin field (August, 2012-January, 2014), shoreline on the north and south of the port appears to move towards attaining stability.

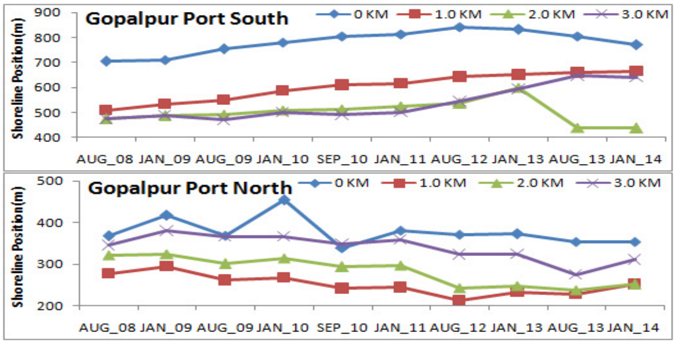

Fig. 2. Changes in Shoreline positions along South and North of Gopalpur Port during August, 2008 to January, 2014.

Fig. 3 depicts the long term (1978-2014) shoreline change rate in the pre (1978-2008) and post (2008-2014) construction phases. Shoreline change rate on the north of the port is negative (0-1.06m/yr) in the pre construction phase while it is enhanced (0.55-4.01m/yr) in the post construction phase. It is also observed that there is positive shoreline change rate (0.08-0.98m/yr) for some portion of the extreme north beach in the post construction phase. Shoreline change rate on the south of the port is positive (deposition) (0-3.23m/yr) for the portion of the beach from SG to 2km south (Fig.1b) and negative (erosion) (0-0.72m/yr) for the south beach including Gopalpur tourist beach in the pre construction phase. However, in the post construction phase the shoreline change rate in the entire south beach including Gopalpur tourist beach is positive (deposition) and the rate (0.66-14.29m/yr) is much higher than that of pre construction phase. This higher positive shoreline change rate on the south beach is attributed to the construction of groins and breakwaters and corroborates with our observed results (Fig. 2).

Mishra et al. (2014) studied the nearshore processes at Gopalpur port through numerical model simulation and its calibration with field measurements. The study revealed that the sediment transport rate (m^3/s) is relatively higher during monsoon (June, July, August and September) as compared to other seasons and is associated with relatively higher breaking wave height (m) and current (m/s). Shoreline change prediction with the existing coastal structures indicated that coastline retreat could be about 70m on the north of the port by the end of 2017. The study suggested that beach nourishment of 3111 m^3/day per km length of the coast shall stabilize the beach by 2017.

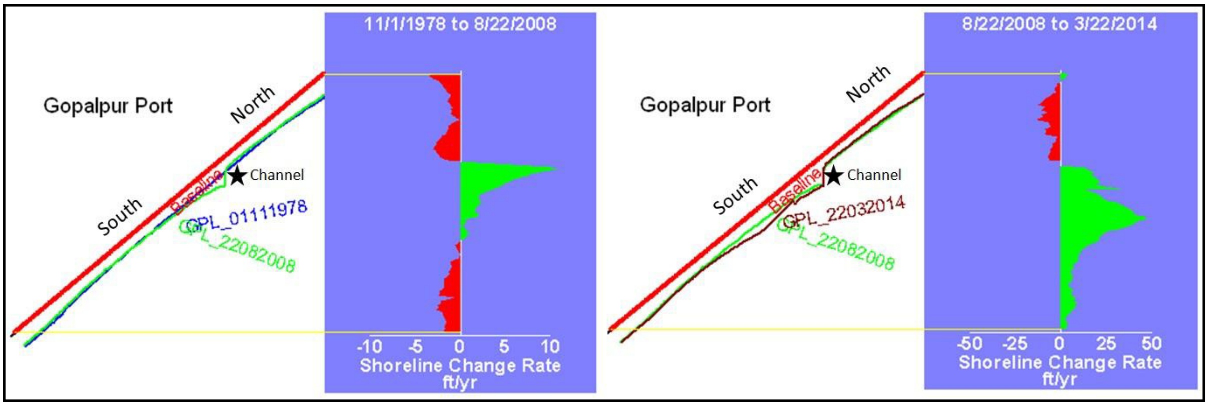

Fig 3. Shoreline Change Rate (ft/yr) at south and north of Gopalpur Port during 1978-2014.

3.1.3 Beach Width and Volume

Table 1 depicts the change in beach width and beach volume at four transects on the south (GPLS_1 to GPLS_4) and four on the north (GPLN_1 to GPLN_4) of Gopalpur port. The distance between each transect is 500m. The beach to the south of SG shows a predominantly depositional environment (positive change in beach width and volume) from GPLS_1 to GPLS_3 during the first phase (Aug. 08 to Jan. 2011) of our observation, i.e. prior to the construction of breakwaters and northern groin field. However, after the construction of breakwaters and groin field (second phase: Aug, 2012 to Jan. 2014) beach width and volume changes are predominantly negative and occasionally positive. At GPLS_4, located 1.5km south of SG and within the two breakwaters, the changes in beach width and volume are negligible during first phase and predominantly negative during second phase. Beach profiles on the south indicate stable berms on the backshore, large ridge on the foreshore, which are prograding in nature, migrates towards offshore continuously.

Table 1. Changes in beach width and volume near Gopalpur Port South and North during August, 2008 to January, 2014.

	Change in Beach Width (m)				**Change in Beach Volume (m)**			
PORT SOUTH	**GPLS-1**	**GPLS-2**	**GPLS-3**	**GPLS-4**	**GPLS-1**	**GPLS-2**	**GPLS-3**	**GPLS-4**
Aug'08-Jan'09	34.35	59.16	37.24	-18.65	93.95	122.45	63.76	-140.31
Jan'09-Aug'09	24.52	27.82	19.04	11.53	59.28	109.84	75.77	61.10
Aug'09-Jan'10	30.77	21.68	21.71	9.70	62.39	29.47	31.51	1.16
Jan'10-Aug'10	40.07	16.24	21.08	18.34	99.36	42.03	83.36	94.30
Aug'10-Jan'11	13.24	24.63	24.49	0.07	-5.47	33.73	57.89	-11.72
Aug'12-Jan'13	4.95	4.95	-2.17	-0.57	-12.87	-12.87	-77.13	-23.54
Jan'13-Aug'13	-66.37	-66.37	-26.34	-132.35	-37.60	-37.60	50.24	-338.95
Aug'13-Jan'14	37.68	37.68	26.80	NA	-19.69	-19.69	-40.94	NA
PORT NORTH	**GPLN-1**	**GPLN-2**	**GPLN-3**	**GPLN-4**	**GPLN-1**	**GPLN-2**	**GPLN-3**	**GPLN-4**
Aug'08-Jan'09	38.29	13.40	14.78	-13.66	44.13	30.21	38.70	-16.47
Jan'09-Aug'09	-33.57	-27.80	-12.52	-9.46	-34.83	-116.15	-24.41	-6.61
Aug'09-Jan'10	75.17	-10.09	-12.09	-3.95	73.87	-142.22	-124.01	-49.22
Jan'10-Aug'10	-101.54	-22.77	-18.57	-5.77	-157.81	-190.02	-130.52	6.88
Aug'10-Jan'11	20.80	-2.88	-3.03	-6.08	8.91	-30.59	-6.35	-52.09
Aug'12-Jan'13	-11.64	-18.40	5.79	10.21	-3.18	-32.08	86.11	-6.31
Jan'13-Aug'13	-5.84	14.73	3.59	-22.37	-13.89	23.44	43.55	-153.25
Aug'13-Jan'14	-2.88	20.85	12.10	26.95	-10.41	237.23	19.19	160.08

The beach to the north of the northern groin is narrow and the beach width and volume changes are predominantly negative after the initial phase (Aug. 08 to Jan. 09) of positive change. In the second phase, after the construction of northern groin field and southern breakwaters, reduction in the erosion trend is observed. Northern beach profiles indicate steep slope at the backshore, no stable berms, absence of ridge on the foreshore. The results suggest the impact of both north and south groins on the downdrift erosion, which to some extent is abated by the construction of northern groin field. The rate of deposition on the south far exceeds the rate of erosion on the north of the port due to trapping of round the year northward longshore transport on the updrift side and preventing the availability of sand on the downdrift side. Therefore, it is suggested that in order to arrest the erosion on the north of the port, beach fill and periodic beach nourishment should be favoured both within and beyond the groin field.

3.2 Paradip Port

Paradip port (Fig. 1c) is located south of the river Mahanadi and is bounded between latitudes $20^{o}\ 30^{'}$ N and $20^{o}\ 45^{'}$ N and longitudes $86^{o}\ 15^{'}$ E and $86^{o}\ 45^{'}$ E. Paradip is a major and oldest port along Odisha coast and commenced its operation during April, 1966. Because of the severe erosion, north side is protected by seawall which is about 2km in length extending up to the Mahanadi river mouth. There is no beach on the north. On the other hand, the beach on the south is well developed.

3.2.1 Shoreline Change

Fig. 4a depicts the observed shoreline positions along south of Paradip port from 0km (refers to PPS_1 in Fig. 1c) to 2km further south. It is observed that the shoreline is narrowest at 0km and has receded significantly between 0-1km during May, 2011 to September, 2011 due to sand mining near the port boundary (PPS_1). However, the shoreline from 1.5 to 2km further south which includes the tourist beach has increased. It is distinct that shoreline gain is maximum during winter as compared to other seasons and is attributed to the low energy wave action during winter. Fig. 4b depicts the shoreline change rate (ft/yr) after the construction of port (1975-2013). The negative shoreline change rates (0-1.65m/yr) on the north of the port and on the extreme south (0-0.97m/yr) are less as compared to the significant positive shoreline change rate (0.09-4.15m/yr) on the south of the port.

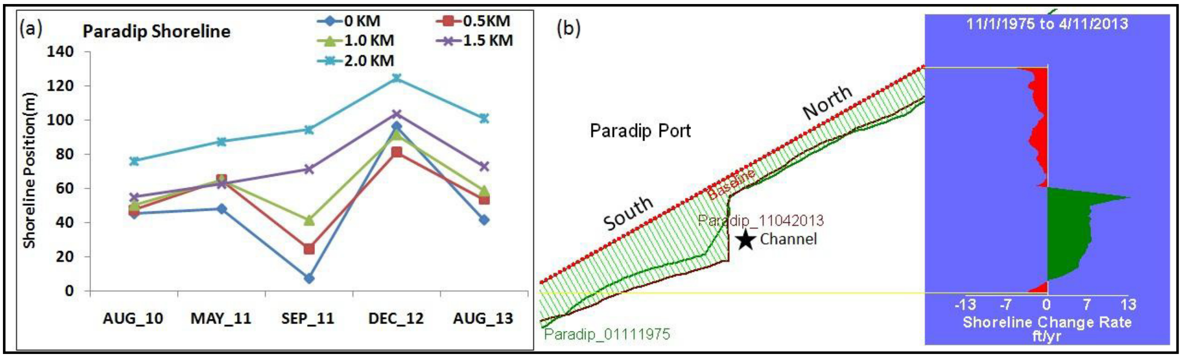

Fig. 4. (a) Shoreline positions along South of Paradip Port during August, 2010 to August, 2013. (b)Shoreline Change Rate (ft/yr) at south and north of Paradip Port during 1975-2013.

3.2.2 Beach Width and Volume

Table 2 depicts the change in beach width and volume at six transects (PPS-1 to PPS-6, Figure 1c) south of the port. PPS_1 and PPS_2 beach width changes are positive during monsoon to winter (September, 2011-December, 2012) and negative during rest of the seasons. At other transects the beach width changes are positive for all the observations but with relatively higher values during monsoon to winter. Beach volume changes during monsoon to winter are predominantly positive for all transects while during other periods volume changes are predominantly negative and occasionally positive. Negative changes in beach width and volume (PPS_1 and PPS_2) very near to the southern breakwater are attributed to the sand mining which violates the coastal regulation zone (CRZ) laws and can be avoided through proper implementation of CRZ laws.

Table 2. Seasonal changes in beach width and volume near south of Paradip Port during August, 2010 to August, 2013.

	Beach Width (m)				Volume (m3/m)			
	Aug'10-May'11	**May'11-Sep'11**	**Sep'11-Dec'12**	**Dec'12-Aug'13**	**Aug'10-May'11**	**May'11-Sep'11**	**Sep'11-Dec'12**	**Dec'12-Aug'13**
PPS-1	-4.42	-29.66	26.67	-4.15	-34.23	-54.09	42.89	22.32
PPS-2	-6.60	-15.40	13.82	-3.28	-26.06	-22.88	31.51	8.53
PPS-3	11.45	2.93	30.40	7.29	35.74	2.36	24.89	24.99
PPS-4	7.30	7.92	27.20	4.86	-20.54	-2.78	20.60	-3.63
PPS-5	8.60	24.38	27.49	9.83	-1.81	50.53	49.16	50.20
PPS-6	6.07	13.26	32.31	12.04	-2.79	38.51	36.30	23.63

3.3 Dhamara Port

Dhamara Port (Fig .1d) is located at latitude of $20^0$47'30" N and longitude $86^0$57'35" E. Phase 1 construction commenced in March, 2007 and the port began commercial operation in May, 2011 with two fully mechanized berths of 350 meters each. The new port is developed in an estuarine environment and is very close the world famous turtle nesting beach at Gahirmatha (Ekakulanasi beach, Fig. 1d). Hence it is very important to assess the shoreline change rate in the pre and post construction phases of the port.

3.3.1 Shoreline Change

Figure 5 depicts the shoreline change rate at Dhamara port during pre (1979-2006) and post (2006-2014) construction phases. In the pre construction phase, the shoreline change rate is distinctly negative (1.14-2.27m/yr) near the present port site and north of it while it is positive (0.21-3.52m/yr) on port south. However, in the post construction phase the scenario has changed drastically with significantly higher positive shoreline change rate (0.46-40.09m/yr) near the port site as well as north of it and almost negligible/no shoreline change rate on the south of the port. Port south includes mudflat and river bank of Dhamara River. Keeping the above changes in view we examined the shoreline change at Ekakulanasi turtle nesting beach, located south of the port (Fig. 1d), during May and August 2010. It is observed that 2745.43 m length and 6220.16m perimeter of the sand spit during May reduced to 2683.53 m and 6080.87m respectively during August. Similarly, the area of the sand spit also reduced from 1004,057.57 sq. m during May to 955,818.61 sq m during August. The reduction in length, perimeter and area of the sand spit is attributed to the high energy monsoonal wave. However, a long term study involving a few seasonal/annual cycles would reveal the erosion/deposition trend of the sand spit. From conservation point of view and considering the world famous status of Ekakulanasi turtle nesting beach, such studies assumes importance.

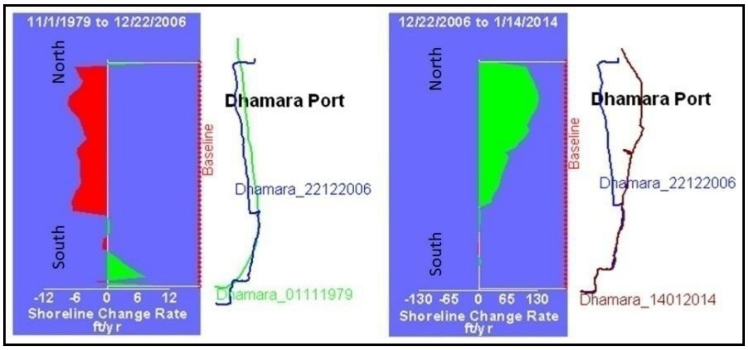

Fig. 5. Shoreline Change Rate (ft/yr) at south and north of Dhamara Port during 1979-2014.

4. Conclusion

Shore perpendicular structures near the port modify round the year unidirectional (towards north) longshore transport along Odisha coast and result in the accumulation of sand on the updrift side (south of port), and erosion on the downdrift (north of port) side. Shoreline changes due to the coastal structures associated with the three port are apparent. However, the nature and magnitude of shoreline change are different from one port to the other. Shoreline movement is distinctly towards sea (deposition) on the south and towards land (erosion) on the north of Gopalpur Port. The rate of deposition is higher as compared to erosion. Beach width and volume changes also corroborate the above views. Impacts of the two groins (SG and NG) and the impacts after the construction of breakwaters and northern groin field are quite different as the shoreline on the south (north) is depositional (erosional) in the former case and later the shoreline tends towards attaining stability. Shoreline change rate (ft/yr) on the north of the port in the pre and post construction phases is negative, albeit with higher rate in later case. A numerical model study (Mishra et at., 2014) suggests beach nourishment of 3111 m^3/day per km length of the coast on the northern side in order to stabilize the coastline by 2017. Shoreline change rate on the south of the port in the pre construction phase (1978-2008) is both positive and negative while it is distinctly positive and higher on the post construction phase (2008-2014). At Paradip port, shoreline change from southern breakwater to 1km south is negative due to the sand mining activity while it is positive further south and is also corroborated by the beach width and volume changes. Long term (1975-2013) shoreline change rate is negative (positive) on the north (south) of the port. At Dhamara Port, pre construction phase (1979-2006) indicates negative (positive) shoreline change rate near the present port site and north (south) of the port while in the post construction phase (2006-2014) it is positive and significantly higher in the present port site and port north, and negligible on port south. The study suggests long term seasonal/annual cycle monitoring of Ekakulanasi spit, the world famous turtle nesting site located south of Dhamara Port, to understand the impacts of the port and to formulate appropriate conservation/management strategy for the rookery. It is envisaged that the result of the present study would serve as important input to formulate Integrated Coastal Zone Management (ICZM) plan for Odisha coast.

Acknowledgements

Part of this study was carried out under a collaborative project with Integrated Coastal and Marine Area Management Project Directorate (ICMAM-PD), Chennai and supported by the Ministry of Earth Sciences, Govt. of India. We are thankful to Gopalpur Ports Limited for providing necessary financial support through consultancy project for monitoring the port environment during 2012-2015.

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Disappearing beaches: From humans to Olive Ridleys, none spared in Odisha

Sea erosion threatens 318 villages in six coastal districts of Ganjam, Puri, Kendrapara, Balasore, Jagatsinghpur and Bhadrak

Published: 21st January 2023 06:53 AM | Last Updated: 21st January 2023 08:03 AM

Though Odisha typically has a pro-gradation coast due to the disposal of sediment and discharge from Mahanadi river, it is now turning into a hotspot of coastal erosion. (Photo | EPS)

By Hemant Kumar Rout

Express News Service

BHUBANESWAR: Jaga Rao of Ramayapatna village in Odisha's Ganjam district used to be a happy man. Like many other traditional fishermen of the region, every morning, he loved setting out into the Bay of Bengal where landing fish was never a problem. Now it is a never-ending struggle with the sea.

The Bay of Bengal has advanced more than 600 metre into his village, located under Chikiti block, gobbling up hundreds of acres of farmland and more than 100 dwelling units. Rao lost his house to the marauding sea but cannot leave Ramayapatna as fishing sustains his five-member family.

Five hundred metres from the new shoreline, he currently lives in the house of another villager who left for Andhra Pradesh in search of work. “The monsoon season brings fear. No one knows when the hungry sea will devour this house too,” he says.

Rao is caught between the devil and the deep sea.

Though Odisha typically has a pro-gradation coast due to the disposal of sediment and discharge from Mahanadi river, it is now turning into a hotspot of coastal erosion with climate change doubling down the impact. The conversion process is rampant in hotspots like Boxipalli and Podampeta in Ganjam district, Baliapanda, Chadrabhaga beach in Puri district, Pentha and Satabhaya in Kendrapara and Chandipur beach and Subarnarekha estuary in Balasore.

At least four villages in Ganjam district face extreme coastal erosion. Podampeta under Ganjam block is now a ghost village as nearly 450 families have already been relocated to a nearby habitation.

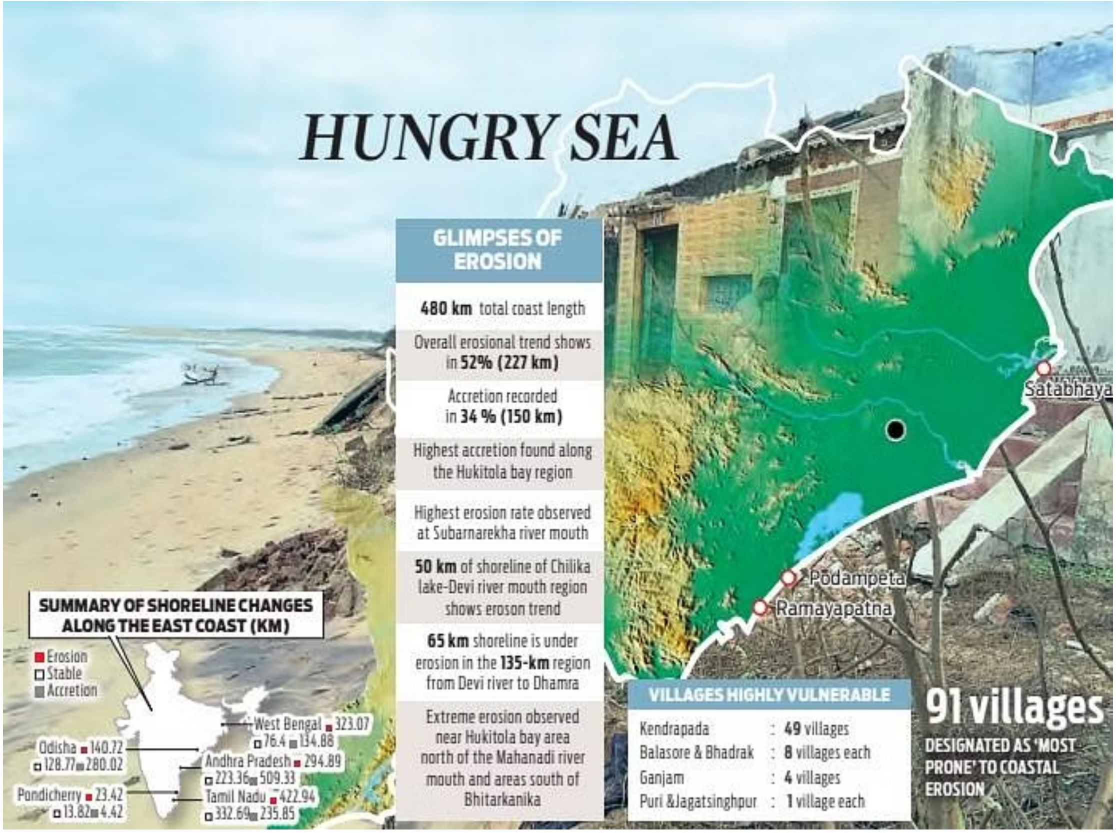
Spectre over six districts

A happy hunting ground for natural calamities, Odisha faces the threat of tropical cyclones like no other. Add to it the misery of vanishing landmasses. State’s Forest, Environment and Climate Change assessed sea erosion threats in 318 villages of six coastal districts - Ganjam, Puri, Kendrapara, Balasore, Jagatsinghpur and Bhadrak. While 91 villages are designated ‘most prone,’ 85 villages are ‘prone’ to sea erosion. With 49 villages facing the charging sea, Kendrapara is the worst hit. Eight each in Balasore and Bhadrak districts, four in Ganjam and one each in Puri and Jagatsinghpur districts are vulnerable to coastal erosion.

ALSO READ | Disappearing beaches: Loss of a crucial buffer along coast hurting Kerala

In Kendrapara district, Satabhaya panchayat under Rajnagar block stands testimony to the erosion as the sea has swallowed it. The panchayat having a cluster of 16 seaside villages has been reduced to a few hamlets by now.

The Subarnarekha river in Balasore district has already swallowed about 75 per cent of Badakhanpur village and its neighbouring Sanakhanpur. Of more than 120 families, only fifty hold on to their dwelling units while the rest have shifted base to neighbouring villages. The ferocious nature has redrawn the map of over 40 villages in Bhogarai, Jaleswar, Baliapal, Remuna and Sadar blocks in the coastal district.



According to a global study conducted by 11 researchers from six universities, almost the entire Gopalpur shoreline experienced erosion between 2010 and 2020 and the construction of Gopalpur port markedly impacted the shoreline dynamics. Of the 480 km sea coast of Odisha, erosion trend has surfaced across 227 km, accretion recorded in 150 km while coast character remains unchanged over the rest.

Professor of Geography department at FM University Manoranjan Mishra said most ports on the east coast have shown a pattern of erosion and deposition, 52 per cent Odisha coastline faces erosion at one stretch or other. Construction of breakwater perpendicular to the coastline prevented the uniform distribution of littoral sediments in all cases, he said.

Coastal infrastructure

With the Odisha government contemplating construction of 14 more ports, researchers warn such infrastructure could potentially impact its fragile coastal system. Resilience of the coastal landforms is compromised due to the construction of onshore and offshore coastal infrastructures in order to meet the growing demand for economic activities.

Previous studies too had observed a similar trend of erosion and accretion along the eastern coast. The northern part of Paradip port experiences intense erosion due to the construction of a rigid engineering structure (sea wall). The southern part of the port also had signs of erosion. Although accretion pattern was witnessed along Dhamra port from 1990 to 2000, drastic erosion was observed after the port area development in 2007.

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Meanwhile, closer to Ramayapatna, erosion has impacted nesting of Olive Ridley turtles. The impact was so high that turtles for the first time in 30 years changed their nesting site from the Rushikulya rookery to a nearby Island, south of Rushikulya river, said secretary of Rushikulya Sea Turtle Protection Committee Rabindranath Sahu. Earlier, the marine turtles used to lay eggs on the 5 km beach from Purunabandha to Podampeta, north of the river mouth.

While state government had relocated residents from Satabhaya and Podampeta and laid geo-synthetic walls along the Satabhaya coast as part of protection measures, the Ganjam seaside villages have long been demanding a protection wall.

Superintendent engineer, Berhampur Irrigation division Jayadeep Panda said as per recommendations made by National Institute of Oceanography (NIO), a 3.4 metre sea wall along the 1200 metre stretch covering Ramayapatna will be built at a cost of `23 crore. “Work will start by next month and it will provide protection to the villages south of Rushikulya. The Podampeta stretch will be taken up in the next phase,” he said.

“The best way forward is to protect salt marshes and mud flats in the coastal regions. Besides, mangrove protection and rejuvenation must be focussed. Similarly, it is critical to create soft structure instead of hard structures to fend off the impact of erosion," said PCCF and Project Director, Integrated Coastal Zone Management Project Susanta Nanda.

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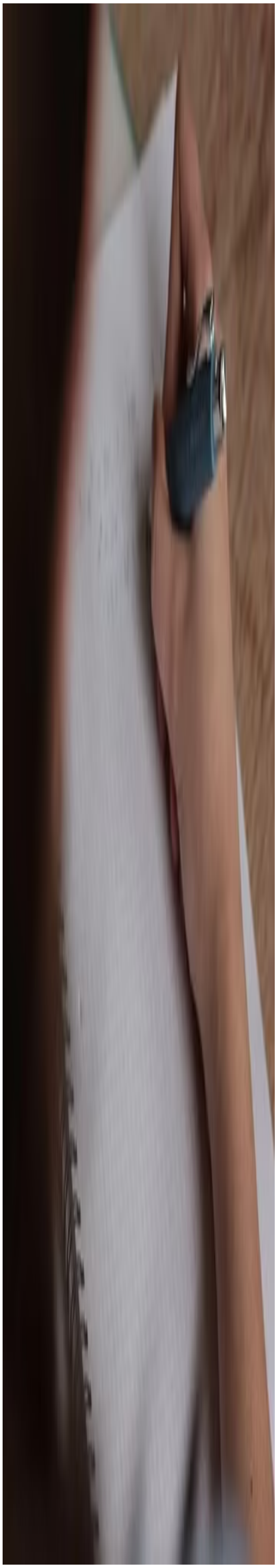

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