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Ecoengineering with Ecohydrology: successes and failures in estuarine restoration

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Abstract

Ecological Engineering (or Ecoengineering) is increasingly used in estuaries to re-create and restore ecosystems degraded by human activities, including reduced water flow or land poldered for agricultural use. Here we focus on ecosystem recolonization by the biota and their functioning and we separate *Type A Ecoengineering* where the physico-chemical structure is modified on the basis that ecological structure and functioning will then follow, and *Type B Ecoengineering* where the biota are engineered directly such as through restocking or replanting. Modifying the physical system to create and restore natural processes and habitats relies on successfully applying Ecohydrology, where suitable physical conditions, especially hydrography and sedimentology, are created to recover estuarine ecology by natural or human-mediated colonisation of primary producers and consumers, or habitat creation. This successional process then allows wading birds and fish to reoccupy the rehabilitated areas, thus restoring the natural food web and recreating nursery areas for aquatic biota. We describe Ecohydrology principles applied during Ecoengineering restoration projects in Europe, Australia, Asia, South Africa and North America. These show some successful and sustainable approaches but also others that were less than successful and not sustainable despite the best of intentions (and which may even have harmed the ecology). Some schemes may be 'good for the ecologists', as conservationists consider it successful that at least some habitat was created, albeit in the short-term, but arguably did little for the overall ecology of the area in space or time. We indicate the trade-offs between the short- and long-term value of restored and created ecosystems, the success at developing natural structure and functioning in disturbed estuaries, the role of this in estuarine and wetland management, and the costs

and benefits of Ecoengineering to the socio-ecological system. These global case studies provide important lessons for both the science and management of estuaries, including that successful estuarine restoration is a complex and often difficult process, and that Ecoengineering with Ecohydrology aims to control and/or simulate natural ecosystem processes.

Keywords: estuarine; ecoengineering; ecohydrology; managed realignment; estuarine processes; habitat restoration.

Introduction

Background and Definitions

Environmental management aims to fulfil the 'big idea', i.e. 'to protect and enhance the natural structure and functioning of the ecosystem while at the same time ensuring the processes which deliver ecosystem services from which we then obtain societal goods and benefits' (Elliott, 2014). This is also the raison d'être of ecological engineering which aims to restore the desired ecosystem functioning but, as we emphasise here, using Ecohydrology. The main physical processes behind the restoration, recovery or maintenance of the ecology of systems based on management actions is Ecohydrology (Wolanski & Elliott, 2015), and may be regarded as the means of achieving these end-points (Box 1). Ecological engineering (or Ecoengineering) is widely regarded as engineering the physico-chemical processes, including water quality and quantity, to improve the ecology (what we term Type A) but it also includes engineering the ecology (e.g. by replanting, restocking, etc) (Type B). This review emphasises Type A Ecoengineering initiatives which lead to the recolonization of biota and their food web relationships but, because of space restrictions, gives less attention to Type B ones involving the active introductions of organisms.

Bergen et al. (2001) considered that there are five design principles which inform ecological engineering. Modifying the first two of these slightly: (1) ecohydrological principles should be used to ensure an appropriate, natural suitable and sustainable physico-chemical system, and (2) the design should encompass local features and so be site-specific. The remaining principles are that the design parameters and features should (3) be kept simple in order to deliver the functioning required but with the simplest design; (4) use energy inside the system or, if coming from outside then work with nature, such as existing flow conditions, and lastly (5) aid the natural system and help achieve social goals and thus have an ethical dimension; this may involve 'over-engineering' the design in order to further protect human safety and property. These principles therefore aim to produce at least a 'win-win' for economy and ecology or even 'triple wins' by including human safety.

Ecoengineering may involve 'hard' or 'soft' engineering solutions to rehabilitate estuarine systems. The former encompasses permanent physical features (e.g., concrete groynes) whereas the latter involves temporary or 'soft' features (e.g., substratum modification, such as by dredging or beach nourishment) in rehabilitation. As we aim to show, these always involve trade-offs, often the underlying conundrum of Ecoengineering, i.e. benefits to safety and economy may only produce a ‘feel good’ benefit for society in general and ecologists in particular without fully restoring the ecology of the natural environment.

Ecoengineering is therefore regarded here as manipulating the estuarine or coastal system either to restore it from past degradation or to improve its delivery of nature conservation and natural structure and functioning to increase ecosystem goods, services and societal benefits (Box 2). This may include recovery from the excesses of development designed to achieve societal benefits but often at the expense of the natural system, e.g. poldering for agriculture which removes coastal and estuarine wetlands. While there is the aim for Ecoengineering to achieve wins for ecology and the economy, and management measures are often carried out with the best intentions, this is not always the case. The aims and objectives of the management measures may be poorly defined, thus making it difficult to determine success. Furthermore, a misdiagnosis about how we should attempt to restore nature is often caused by uncertainty in what constitutes a win-win solution using science and engineering (Rodgers, 2000).

Ecoengineering often involves continuous intervention or maintaining management actions, with Ecohydrology providing the underlying principles for Ecoengineering (Box 3). Here we take the view that Ecohydrology often establishes the dynamic processes necessary to meet the aims, while Ecoengineering often aims to produce a required status (such as a restored seagrass bed) rather than restoring all natural dynamic processes (unimpeded water movement, salinity balance, sediment erosion-deposition cycles, etc.).

Box 1 Estuarine Ecohydrology
The science and understanding of the links between the physical functioning and the means by which it creates the appropriate ecological functioning of an estuary. It assumes that the ecology is primarily driven by the physics, which in turn affects the biological processes operating within a system. It includes changing the physiography and manipulating the freshwater flows from the catchment and it is also influenced by the anthropogenic users and uses of the estuary, some of which will have modified and impacted both the physics and the ecology. It is that knowledge which guides the management of the entire river basin from the headwaters down to the coastal zone, which Ecohydrology views as an ecosystem.

Box 2 Estuarine Ecological Engineering
Using Ecohydrology principles and knowledge to modify and achieve ecological and societal aims for an ecosystem, firstly by engineering the physics to produce particularly desirable

niches which in turn lets the ecology and habitats develop, especially if the colonising species are ecological engineers (Type A Ecoengineering). Secondly, by engineering the ecology by restocking or replanting, in turn creating habitats or letting the ecological engineer species modify habitats, thus enhancing the physical-biological links (Type B Ecoengineering). Ecoengineering initiatives often aim to accelerate natural rehabilitation and sometimes harness dynamic variability. However, they often only achieve establishing a static system (the desired state) even if this does not include all natural successional processes and stages.

Box 3 Ecohydrology with Ecoengineering

While Ecohydrology aims to operate across the whole catchment-coast continuum, Ecoengineering usually occurs at a smaller scale and will seldom recreate pristine estuaries given the huge human populations living on their shores, but it aims to create ecosystems with at least some attributes of the original systems. It should be accompanied by regulating certain human activities and is more than just integrated river basin management. Primarily it aims to improve the ecology, and provide benefits for the economy and the safety of society (i.e., so-called triple wins). It aims to redress the balance after adverse historical changes, especially coastal and estuarine wetland removal, without unacceptable environmental trade-offs. Ideally, it provides relatively low-cost technologies for mitigating the impact on estuaries and coasts of human activities throughout the river basin, for using and enhancing the natural capacity of the water bodies to absorb and process excess nutrients and contaminants, and for increasing ecosystem resilience to accommodate global stressors such as climate change. In essence Ecohydrology is the underlying process/abiotic drivers into which Ecoengineering fits and by which Ecoengineering is delivered, i.e. Ecohydrology provides the underlying science and Ecoengineering is the mechanism for creating the ecology.

Following the conceptual model of Elliott et al. (2007), giving ecosystem improvement options from degradation (Fig. 1), habitats can be restored in terms of an improvement on the degraded state but which may not necessarily return to the original state. Recovery can be to the original state or some distance along this or another trajectory of regaining ecosystem quality, based on the societal demand for ecosystem services and/or a perceived good ecological status, but not necessarily leading to the original state (Fig. 2, Aronson and Le Floc'h, 1996; Bullock et al., 2011). While it is almost impossible to restore systems to their original state, restoration and rehabilitation projects should aim to achieve their 'remaining natural potential', given possible irreversible effects of degradation, catchment environmental constraints (e.g., expected future changes due to climate perturbations), socio-economic constraints (e.g. resource availability) and societal support. While we may culturally tend to seek the original state, the 'remaining natural potential' will more than likely result in novel ecosystems (Hobbs et al., 2009, 2013) that we must manage for 'reconciliation ecology' (Rosenzweig 2003). The shifting baselines of the Anthropocene will never allow us to 'Return to Neverland' (Duarte et al., 2009; Elliott et al., 2015; Kopf et al., 2015), but we may be able to restore some coastal ecosystems to the new normal that is represented by regional reference conditions.

Newly created ecosystems, including sites restored to an historical wetland situation, may also deliver highly-valued ecosystem services (see below) (Bullock et al., 2011). Hence, there is the need to take a pragmatic approach to planning and monitoring ecosystem restoration at the national and regional (trans-national) level. This should encompass the different spatio-temporal scales at which Drivers of degradation, Activities, Pressures, State changes, Impacts (on human Welfare) and management Responses (defined often as Measures) operate, (i.e. the DAPSI(W)R(M) framework) (Wolanski and Elliott, 2015).

Successful restoration needs to be judged against SMART (Specific, Measurable, Achievable, Relevant/Realistic and Time-bounded) objectives and whether the 'achievable' ecosystem state/integrity satisfactorily delivers ecosystem services (Turner and Schaafsma, 2015). Ecosystem management and hence restoration is essentially Risk Analysis and Risk Management, i.e., analysing the risks of ecosystems being degraded and the risks of management measures not achieving the desired improvement (Cormier et al., 2013). Mitigation and habitat compensation measures can then be used to either minimise or offset habitat loss and ecosystem degradation. In Fig. 1, each of the curved arrows leading from the degraded situation, together with the management measures of compensation and habitat creation, all require Ecoengineering even though it may be just to remove the stressor.

This review links Ecohydrology to Ecoengineering to address the above ecological and socio-economic imperatives. It uses case studies to focus on Type A Ecoengineering, the engineering of the physical habitat hopefully to produce conditions suitable for colonisation by the biota, but also provides examples of Type B Ecoengineering in which the biota are directly engineered through restocking or replanting; the latter in itself often then alters the physical aspects. This review aims to show the Ecoengineering successes but also the problems, some of which may even harm the ecology, through a background based on Ecohydrology practices and principles. In contrast, engineering where the aims are not to improve the ecology are not covered here, for example for flood protection using hard engineering. In many of those cases, the ecology tends to be harmed to varying degrees. Similarly, because of space, the major field of water quality bioremediation using aquatic organisms, e.g. bivalves and seagrasses, is not covered here (Huesemann et al., 2009; Petersen et al., 2014).

The role of Ecohydrology in designing management measures

The essence of Ecohydrology is the role of the environment in influencing organisms and vice versa – of the organisms building, filling and altering niches (Wolanski and Elliott, 2015). The hydrophysical regime creates the abiotic environment (the sediment and water fundamental niches) which is then colonised by biota and in turn the colonising organisms interact to modify the system (Gray and Elliott, 2009). Ecohydrology therefore has three interrelated and consecutive aspects required for integrated water body (e.g., an estuary), river basin or catchment management. Firstly, the hydrological processes at the management scale (either as an estuary, lagoon, water body or catchment); secondly, the ecological structure and functioning at the relevant spatial and temporal scales and its relation to the ecological carrying capacity (Strong et al., 2015); thirdly, in turn, the socio-ecological system, the production of ecosystem services that can be used to deliver societal goods and benefits (Luisetti et al., 2014, Turner and Schaafsma, 2015). This can then be phrased as testable hypotheses (from Wolanski and Elliott, 2015):

- H1: hydrological processes generally determine and initially regulate the structure of the biotic communities;
- H2: ecological functioning (rate-processes) arises from the interactions between the elements of ecological structure (such as individuals, populations and communities);
- H3: the biotic structure and functioning will induce feedback loops which can then help to structure the physico-chemical environment (such as removing nutrients or changing water flows);
- H4: the previous three hypotheses can then be integrated with management measures, such as water control (e.g., compensation flows) or hydro-technical infrastructure (e.g., barriers), to achieve sustainable water management, the protection of ecosystem services and the delivery of societal goods and benefits.

This understanding of structure and functioning then informs what measures should be taken and where, how and why, especially working with rather than against nature. Based on understanding the effects of human activities, links can then be made between the hydrophysical structure and processes and their effect on structuring ecosystems together with their goods and services. Engineering and ecosystem modification mechanisms can then be used to improve or restore ecology and fulfil societal needs. Table 1 lists measures to prevent a deterioration of an estuary or to remediate this after it has occurred, grouped according to hydrological, morphological, biological, chemical and physical techniques/technologies. Those measures may be either sanctioned or required by governance (laws, policies, etc.) and should fulfil the 10-tenets of sustainability: ecologically sustainable, economically viable, technologically feasible, socially desirable/tolerable, legally permissible, administratively achievable, politically expedient, ethically defensible (morally correct), culturally inclusive and effectively communicable (Elliott, 2013, Barnard and Elliott, 2015).

Case-studies of Ecoengineering with Ecohydrology

Many attempts worldwide to use Ecohydrology for Ecoengineering have had varying degrees of success in restoration, for example, for compensation schemes, creation/re-creation of habitats, hydromorphological modification, remediation in the short term, reversing a historical legacy and changing the nature or societal use of the area. They have been used to supplement the loss of resources/species/habitats/populations and to ecologically enhance a development (e.g., creating shoreline habitat alongside erosion protection). Finally they have been used to increase public safety (e.g. to stop flooding) and to aim for the 'triple-wins' using more cost-effective soft vs hard engineering approaches. Each of these involves decisions on what, where, how and when to engineer – for example, the shape of an area (physiography), water exchange, water balance, sediment supply, bathymetry/topography, water quality (e.g. oxygen), supply of colonizing organisms and habitat suitability for target fauna. Below, we present trade-offs in Ecoengineering using global examples to indicate success or otherwise. The examples show that often restoration projects are driven by the norms and values required by the local or regional community; this may even reflect the political or governance driver for investing in restoration such as required by European Directives (Boyes and Elliott, 2014).

Case-study 1 - The conundrum of Ecoengineering trade-offs in altered (restored/created/re-created) ecosystems

(i) Hydrodynamic interference: Trade-offs between the aims and outcomes of ecosystem restoration are implicit in altered ecosystems, where societal, economic and cultural manipulations tend to inhibit recovery toward pre-disturbance states (Fig. 2, Aronson and Le Floc'h, 1996). Ecoengineering management measures may be motivated by reconstituting natural, pre-existing ecosystems but there often appears to be constraints on the capacity to achieve a return to dynamic natural Ecohydrology processes. This may be our inability to remove historical stressors and let the natural dynamic processes operate, or a reluctance to relinquish ecosystem services and societal goods and benefits that have evolved under the shaping of our now altered ecosystems. Certainly, public safety and protection of human infrastructure using dykes (levees) that now surround many estuaries cannot easily be countered, although even dyke relocation, breaching and sluice modification has become a questionable restoration action in some estuarine deltas (Disco, 2002; Roman and Burdick, 2014; van Staveren et al., 2014), albeit not without considerable inertia (Marks et al., 2014). For example, the increasing interest in 'engineering with nature' still predominantly seeks Ecoengineering to solve the trade-off between flood and storm surge management and restoration of delta dynamics associated ecological benefits (van Staveren et al., 2014).

Such trade-offs are the conundrum of Ecoengineering for coastal restoration in most developed coasts and estuaries where Ecohydrology processes are theoretically recoverable. Restoration trade-offs and constraints that involve both competing aims and values as well as novel ecosystems include modified tide gates, beach nourishment, installation of large structures and the artificial excavation of channels (Table 2).

Tide gates are simple Ecoengineering approaches for agriculture or lowland development to drain accumulated freshwater in wetlands behind dykes (polders), which allows freshwater to drain during low tides but prevent tidal inflow on higher tides and floods. While generally effective in preventing flooding behind dykes, traditional tide gates impact water quality, obstruct fish passage and other ecological connectivity and degrade channel habitat (Vranken and Oenema, 1990; Portnoy, 1999; Kroon and Ansell, 2006). Without constant maintenance, the traditional 'flap' tide gates may be kept open by wood and other debris, thus both decreasing their efficiency and allowing nekton to enter and potentially become trapped in degraded dyked wetland habitat.

Recognising the above impacts on nature, Ecoengineering solutions have ensured tide gates allow greater, bi-directional flow in dyked areas to improve nekton passage and habitat quality (Giannico and Souder, 2004, 2005). These 'fish-friendly' or self-regulating tide gates (SRTG) are a compromise to restoration measures that return the historic wetlands behind dykes to a degree of unimpeded tidal action. While connectivity and water quality by the SRTG is generally improved over the traditional tide gates, the physical and other fish habitat conditions are still relatively poor for estuarine-dependent species such as endangered or threatened fish species in the north-west Pacific region (Greene et al., 2012). Whether such Ecoengineering tide gates will partially attain desired improvements in estuarine wetland connectivity and habitat quality, depends on temporal access and habitat quality requirements of the targeted nekton species versus those more adapted to restricted tidal systems (Boys et al., 2012; Franklin and Hodges, 2015). Similarly, the amount of investment and sustained maintenance are an acceptable compromise, but made more complicated by the often energy-demanding characteristics of SRTG and other 'automatic' tide gates (Glamore, 2012; Reiner, 2012).

As with all trade-offs, the socio-economic benefits of enhancing, rather than restoring, a compromised, created ecosystem for services such as flood protection will need to be balanced against the ecological costs and long-term sustainability of the Ecoengineering solution. With sea level rise, Ecoengineering approaches such as SRTG, "while providing short-term benefits, may be an evolutionary dead-end" (Rozas, 2012). In cases aiming for more restorative Ecohydrology processes, Ecoengineering approaches such as the controlled reduced tidal system (CRTS) have been designed, such as in the Scheldt Estuary, to attempt to replicate the spring-neap tidal cycle and allow flood water storage, producing somewhat modified, early successional tidal marshes (Beauchard et al., 2011).

(ii) Modified morphology: Engineering channel network morphology in estuarine tidal marsh restoration challenges the belief that Ecoengineering restoration designs can effectively reproduce or initiate the desired tidal channel network design and function. Our technical understanding of hydraulic geometry principles and other factors that regulate tidal channel network characteristics, such as width and depth proportions and sinuosity, is sufficient to inform restoration design and even numerical models (Zeff, 1999; Hood, 2002, 2004, 2007a&b, 2014; Williams and Orr, 2002; Williams et al., 2002; Fagherazzi and Sun, 2005; Lawrence et al., 2004; D'Alpaos et al., 2005). However, this knowledge is inconsistently applied (Zeff, 1999; Hood, 2014) and the stochasticity in the other factors influencing channel network development (vegetation, soil compaction, microtopography, subsequent landscape change) make uncertain the form and rate of channel network development. Furthermore, where the aim for channel network creation is associated with dyke removal, subsidence behind the dykes will probably cause channel network evolution under different erosional processes than the transgressive migration that originally formed the pre-development network. This questions the applicability of the historic network form as an assessment benchmark.

Ecoengineering approaches can be *emulative*, where precise tidal geometry principles are applied to design the endpoint morphology (Zeff, 1999), or *adaptive* in which simple channels are constructed and expected to eventually adjust to a more natural cross-section equilibrium (Rozas, 2012). Restoration practitioners often distrust substratum erosion processes to produce natural channel development. Despite this, the perceived benefit of initial tidal channel habitat for fish and wildlife and other functions, including aesthetic considerations, typically drives excavation even without a cost-benefit analysis, both in terms of the excavation itself and the time required to restructure the Ecohydrology characteristics. The cost, benefit and uncertainty trade-offs have yet to be evaluated between the active Ecoengineering high channel development rate and high excavation costs, versus passive Ecohydrology-based lower channel development with no excavation costs (Hood, 2006).

(iii) Introduced structures: Large wood debris is recognized as an important structuring feature in aquatic ecosystems, especially in large, temperate rainforest rivers (Gonor et al., 1988; Maser and Sedell, 1994). While often a prominent feature, the function of large wood in coastal ecosystems is less clear and may be particularly dependent on landscape context (Simenstad et al., 2003). However, large wood deliberately placed in estuarine wetland and along beaches can provide various ecosystem functions, such as shaping beach morphology, reducing shoreline erosion, enhancing biodiversity and providing important habitat for a diverse biota (Everett and Ruiz, 1993; Kennedy and Woods, 2012).

As with other Ecohydrology-associated change, the decline in natural quality, input and transport of large wood into estuaries and along coasts epitomises the degradation of previous fundamental ecosystem processes (Rich et al., 2014). Nevertheless, considerable attention to the design and cost of both passive Ecohydrology natural recruitment and active Ecoengineering placement schemes in river systems (Shields et al., 2004; Kail and Hering, 2005; Manners and Doyle, 2008) is now also being employed for estuarine and coastal restoration planning (e.g., Heathfield and Walker, 2011). While providing beneficial services in terms of wind-driven wave erosion mediation and ecological functions using tree debris (e.g. habitat and biodiversity enhancement) warrants engineering, the efficacy and collateral ecological functions observed in rivers (Larson et al., 2001; Lester and Boulton, 2008) are often lacking in estuarine and coastal applications.

The Ecohydrology-Ecoengineering trade-off in the case of large wood debris is analogous to tide gate enhancement and beach nourishment in that the more sustainable restoration of wood debris is often not feasible in most created ecosystems where natural catchment forests have been destroyed. It is also prohibitively expensive to address the underlying degradation of Ecohydrology processes, even though this would produce a more sustainable and resilient solution. Furthermore, the dynamic movement of wood debris in estuaries and beaches that accounts for much of its ecological function is often not envisaged for various reasons, some of which are socio-cultural (Piégay et al., 2005).

(iv) Obstructing Sediment Delivery: Beach nourishment is an exemplar of an Ecoengineering trade-off. Created ecosystems have emerged along estuarine and coastal shorelines where diverse soft or hard engineering infrastructure (e,g, groynes, bulkheads) inhibit or prohibit long-shore transport of sediments that, under natural hydrological processes, would continue to ‘feed’ these beaches (Nordstrom, 2014). In more extreme cases, sediment delivery to estuarine and coastal shorelines is trapped behind large and multiple dams in catchments. In addition, although the net effects of accelerated climate change are uncertain, they are likely to involve increased coastal erosion (Elliott et al., 2015).

Beach nourishment, the artificial deposition of comparable substrata, is a long-standing Ecoengineering compromise to fully restore sediment transport processes and often used to keep pace with sea level rise, thus preventing erosion (Nordstrom, 2005; Speybroeck et al., 2006). Ecological benefits of beach nourishment are almost incidental on classically faunistically poor open sedimentary coastlines, although there may be more ecological benefits in estuaries. As most Ecohydrology processes of sediment transport still operate in the absence of, or are limited by sediment delivery, one cycle of beach nourishment to counter erosion is not a sustainable management measure in sediment-starved settings. Thus, the trade-off is tenuous if the process of sediment delivery cannot be restored, and nourished beaches will continue to undergo episodic changes in profiles, sediment structure, and habitat quality for intertidal species. Beach nourishment can impact fauna over both the short (Schlacher et al., 2012) and long term (Manzanera et al., 2014) but these responses are typically not assessed or included in cost-benefit analyses of beach nourishment management planning (Martino and Amos, 2015).

Case-study 2 - Managed realignment, e.g. NW Europe, Humber Estuary (UK)

Coastal flood protection in estuaries is driven by three factors: public safety, economy and environment. Whilst the first is paramount (i.e., protection of housing and industries), economics drive the long-term strategy for coastal flood protection in less populated areas (e.g., cost-effective schemes) and the type of engineering measures required (e.g., hard or soft flood defences). Ecology has been the focus of environmental legislation and there is increasing public awareness of environmental issues such as sea level rise caused by climate change. The Humber Estuary, Eastern England, has >400 000 people living below the high water mark, as well as industries and an internationally important port complex (Fig. 4a). The intertidal areas of the Humber Estuary are also of importance for fish and waterbirds and their prey, with the habitat causing the estuary to be protected under national, European and international designations.

With a network of 230 km of flood defences in need of improvements, the Environment Agency (EA) policy is to retreat the line of defences where improvements were not cost-effective (e.g., failing seawalls backed by farmland). Setting back the defence, known as managed realignment or de-polderisation, at two sites (Paull Holme Strays and Alkborough Flats, Fig. 4a), is part of the Humber Estuary Flood Risk Management Strategy. This aims to increase the flood storage capacity of the estuary, provide compensatory habitats for direct flood defence schemes, and accommodate future coastal squeeze losses due to flood defence work improvements (Environment Agency, 2008) (Table 3). Two other realignment sites (Chowder Ness and Welwick, Fig. 4a) on the Humber are directly linked to compensation for wetland loss through port developments.

Ecohydrological principles were applied to realignment sites whereby physical processes (i.e., tidal inundation and accretion of sediment) and in turn their control on other ecological processes (i.e., colonization by invertebrates, fish and birds) aimed to restore and re-create functioning intertidal areas. The ecological development of Paull Holme Strays and Alkborough Flats was monitored over a 10 year period to reveal, at Paull Holme Stray (PHS), a rapid evolution of intertidal habitat from the previous farmland due to high sediment deposition. From the breaching of the site in 2003, the sediment surface level in the realignment site accreted between 8.1 cm at the highest elevation (3.15 m OD) and 118 cm at the lowest elevation (2.78 m OD) by 2013 (Brown, 2014). Within three years, invertebrates and foraging waterbird communities had rapidly colonised the site and showed some similarities with those found on a contiguous established and reference intertidal area (Mander et al., 2007; Mazik et al., 2007). However, the position of the PHS realignment site high in the tidal frame (between 2.78 m and 3.65 m above OD in 2013) and its configuration (Fig. 3), resulted in increasing elevation over time which was detrimental to the desired aim of creating a lagoon; this was due to a deficiency in the modelling carried out pre-breach. Vegetation cover increased each year and saltmarsh vegetation cover was approx. 36% in 2007 (five years after breaching), increasing to 58% in 2013 (10 years after breaching) (Brown, 2014).

Similarly in the Humber Estuary, the Chowder Ness Managed Realignment site was created as compensation for the loss of a middle-estuary mudflat area due to port development at Immingham Harbour. Considerable attention was paid to site preparation, with new seawalls, sculpting the topography and removal of previous vegetation (Fig. 4a, b). This produced early recovery such that within a year an apparently fully-functioning mudflat had been created, supporting typical sediment, infauna and predators such as wading birds (Fig. 4c). However, continued accretion and the lack of any maintenance allowed the system to progress through a mudflat habitat and into low and then high marsh (Fig. 4d), thereby negating the value as a compensation site for the loss of mudflat. This sequence, in passing through the desired to an undesired ecological state, appears to be the result of either unreasonable expectations, poor initially-defined objectives and/or poor science involving poorly predicted sediment loadings.

Maintaining the compensatory habitats targeted (e.g., wet mud supporting invertebrates and foraging waterbirds) and hence enhancing the estuarine carrying capacity is challenging. Ecoengineering techniques such as realignment are required to maintain desired habitats such as mudflats, e.g., by reducing the elevation of the sediment surface. Elevation regulates both physical (e.g., tidal inundation and sediment accretion, Garbutt et al., 2006) and biological parameters (e.g., benthic macrofauna abundance and biomass, Mazik et al., 2010) and so managed realignment may offset the long-term loss of intertidal areas due to rising sea level as this allows the intertidal areas to roll landward. As shown here, the technique is, however, questionable when used by estuarine managers as a compensation site, for example, to replace like-for-like intertidal mudflat, especially when low-shore areas of intertidal mudflat will be lost and replaced by a realignment site on the high shore. Again this shows poorly defined objectives, poor Ecoengineering expectations, and/or poor understanding of Ecohydrology principles.

Case-study 3 - Peel-Harvey system (Western Australia)

The Peel-Harvey Estuary is the largest in south-western Australia (130 km^2; Fig. 5) and a Ramsar-listed wetland that supports key wildlife communities and acts as a 'stepping-stone' for connecting aquatic fauna across the region (Brearley, 2005; Tulbure et al., 2014). It also provides an iconic example of hyper-eutrophication, decline and a subsequent attempt at ecological remediation through a major hard engineering initiative, the Dawesville Cut. Prior to this Ecoengineering initiative in 1994, the Peel-Harvey system had, since the early 1900s, been heavily modified by various other engineering interventions aimed at increasing human benefit as opposed to ecological benefit. Before European settlement, the low-lying coastal plain formed an extensive, interconnected system of wetlands which flooded in winter. As catchment clearing intensified, mainly for agriculture then later for industrial and urban development, so did the construction of major land drainage networks, desnagging and straightening of the waterways and dam building, all of which have removed significant volumes of water from the landscape, reduced freshwater inputs to the estuary and contributed to the loss of many seasonal wetlands (Table 4; Bradby, 1997; Brearley, 2005; Environmental Protection Authority, 2008).

As with many estuaries across south-western Australia, the Peel-Harvey is highly predisposed to degradation from surrounding land development, given its large sandy catchment that readily leaches nutrients, its wide and shallow receiving basins and limited potential for flushing, i.e. a narrow natural entrance channel, tidal range of <0.5 m, extensive freshwater diversion and highly seasonal and diminishing rainfall. Severe eutrophication was first noticed in the 1960s, when extensive macroalgal growths blanketed the Peel Inlet, followed by blooms of the toxic blue-green algae *Nodularia* *spumigena* in the Harvey Estuary from 1978 (Table 4; McComb & Humphries, 1992; Brearley, 2005). While different management approaches were investigated and employed to varying extents (e.g. algal harvesting, voluntary fertilizer management programmes and soil amendments), the scale of the problem required a more immediate solution. Strong community support and political opportunity led ultimately to the construction of a second entrance channel, the Dawesville Cut (2.5 km long, 0.2 km wide and 4-6.5 m deep), to increase tidal flushing of the estuary. It was built at the northern end of the Harvey Estuary, which previously had no direct connection with the sea (Fig. 5), tripling water exchange with the ocean and flushing ~10% of the estuary volume each day (Brearley, 2005).

The Cut was intended as one element of a three-part strategy to improve estuarine condition, together with a catchment management plan to reduce nutrient flows and further algal harvesting. A five-year programme to measure remediation success was also approved by the State Government. However, a consolidated catchment management plan lagged and is still outstanding (Environmental Protection Authority, 2008). Funding for post-Cut monitoring was also criticized as insufficient (Brearley, 2005), and a lack of pre- and/or post-Cut data has hampered understanding of holistic ecosystem effects. Various strategies for improving estuarine water quality have since been developed, including a Water Quality Improvement Plan (WQIP) funded by the State and Commonwealth Governments (Environmental Protection Authority, 2008) aimed at reducing phosphorus (P) delivery through land-use changes. There has also been significant effort towards developing ‘best management practices’ (BMPs) to reduce nutrient flows (e.g. Keipert et al., 2008; Rivers et al., 2013). Despite this, large-scale implementation of these strategies and practices has been limited, and other key issues such as catchment nitrogen flows still require further action.

While the Cut has been successful in achieving its primary objectives of improving estuarine water quality and reducing the most visual signs of ecological decline such as algal blooms, other less visible aspects of the system, most notably the benthos, have not recovered following this Ecoengineering initiative. The system is also now far more influenced by marine conditions and thus while the health of some ecological components has not obviously improved or declined, they are now simply different (Table 4). There are additional challenges, however, in separating the effects of the Cut from those of both climate change and ongoing catchment development. Thus, rainfall and river flows across south-western Australia have fallen by 16 and 50%, respectively, since the 1970s (Silberstein et al., 2012), and population growth in the Peel region (~4.5% pa) is one of the fastest nationally (Australian Bureau of Statistics, 2015).

Given that the Cut aimed to address the main symptoms of eutrophication and was not accompanied by large-scale catchment remediation to address the wider causes, it is not surprising that this Ecoengineering initiative produced only partial ecological improvement. Moreover, not accounting for climate variability, Rivers et al. (2013) predict that if current land management practices continue, then in the next century annual P export to the estuary will be ~9 times that of current levels and ~18 times the target set by the WQIP (Environmental Protection Authority, 2008). They further predict that even with broad-scale implementation of agricultural BMPs, P export over the same period will be at best ~2.5X that of current levels, given the reduced buffering capacity of the system. From an urban perspective, Beckwith & Clement (2013) further caution that while wide-scale adoption of BMPs to reduce fertilizer use can significantly reduce nutrient inputs to the estuary, they are unlikely to achieve the scale of gains sought by the WQIP. Such findings do not imply that catchment measures to reduce nutrient flows are not worthwhile, but highlight the large management challenges in dealing with legacy issues, barriers to positive behavioural change and the pressures of growing coastal populations.

Case-study 4 - St Lucia estuarine system (South Africa)

Lake St Lucia is one of the largest estuarine systems (approximately 35 000 ha) in Africa, part of the iSimangaliso Wetland Park, recognised as a World Heritage Site and a Ramsar Wetland of International Importance and it has even been used as a global review model (Perissinotto et al., 2013). Despite this and its value as a major nursery area for fish and invertebrates, as well as a home to the largest populations of hippopotamus and crocodiles in southern Africa, St Lucia has been subjected to artificial extremes in environmental conditions for more than half a century (Cyrus et al., 2010; Perissinotto et al., 2013). Natural extremes have always been part of its ecology but the environmental fluctuations brought about by Ecoengineering aimed at benefitting sugar farming on the Mfolozi floodplain have compromised the long-term survival of the natural ecosystem (Whitfield and Taylor, 2009).

There have been attempts at remediation since the initial canalization of the Mfolozi Swamps (Table 5). The Ecoengineering associated with the excavation of the Wilson and Warner drains along the Msunduzi and Mfolozi watercourses were the start of a chain of events that lasted more than 80 years and caused major damage to the natural functioning of the St Lucia system (Perissinotto et al., 2013). The past century was also characterized by a belief that engineering solutions could be devised for environmental problems, and St Lucia was no exception. Hence, dredging accumulated sediments, construction of groynes to keep the mouth open, and the excavation of a link canal to bring freshwater to St Lucia, were thought to solve collectively the ecological problems associated with the diversion of the Mfolozi River water away from St Lucia. However, two natural events changed that attitude: firstly, Cyclone Demoina and the destruction of almost all the existing engineering 'solutions' and, secondly, a decade long drought (2002-2012) that almost eradicated the aquatic biota through widespread evaporative loss of surface water and extreme hyperhalinity in the lake (Perissinotto et al., 2013).

In May 2010, a significant, multidisciplinary workshop of St Lucia scientists was held to document available information and research gaps, to assess the implications of reconnecting the Mfolozi River to the St Lucia system and agreed that St Lucia would be unable to survive as a World Heritage Site if the two were not reconnected (Bate et al., 2011). This recommendation was embraced by the iSimangaliso Wetland Authority and much of the artificial sand berm between the Mfolozi and St Lucia systems has now been removed and excavated to facilitate that link. Plans are underway to create additional links between the two systems and to physically remove woody vegetation on the island (created by dredged material) between the two systems. This will allow future episodic floods to scour the accumulated sediment out to sea, thereby fully re-establishing the joint mouth and maybe even the St Lucia 'Bay' that is visible in earlier maps (Whitfield et al., 2013).

Case-study 5 - Richards Bay (South Africa)

The Richards Bay system, on the eastern South African coast, naturally is more of an estuarine lake than an estuarine bay (Whitfield, 1994). The 2 890 ha system (Fig. 9a) originally was only ~0.9 m deep, had salinities ranging from 12-35 during non-river flooding periods, and had a water level ~1 m above mean sea level (Day, 1981). The estuarine lake was connected to the sea by a narrow channel in the north and tidal ranges were ~0.2-0.35 m (Begg, 1978). The system was a popular angling venue and also an important nursery area for both fish and penaeid prawns in KwaZulu-Natal, with >150 invertebrate and fish species being recorded in the *Zostera capensis* beds and mangroves inside the mouth of the estuary alone (Day, 1981).

In 1974, the southern half of the 'bay' was separated from a new harbour development in the north by a berm and a new mouth for the southern 'Sanctuary' area (see below) was created by excavating a channel through the sand dunes to the sea (Fig. 9b). Tidal gates were built into the berm in order to equilibrate water levels between the newly-created Richards Bay harbour and Mhlathuze Estuary (The Sanctuary), since the harbour was calculated to fill quicker with water than the estuary due to its wider and deeper entrance to the sea. It was also envisaged that the gates could be closed in the event of a spate in the Mhlathuze River. However, the flood gates only worked for a short period after commissioning and soon became clogged with sand deposits and then rusted in the closed position.

The harbour engineers were very pleased at being able to 'divert' the high sediment carrying Mhlathuze River into the Sanctuary and this engineering 'solution', in the absence of sediment filtering headwater swamps, caused the Sanctuary area to rapidly accrete with sand and mud that was subsequently colonized by mangroves. Unfortunately, littoral estuarine and freshwater plants in both the new Richards Bay harbour and Sanctuary areas perished due to the increased tidal range and salinities arising from the re-engineered ecosystem (Begg, 1978). The residence time of estuarine water in both compartments was reduced from months to hours, with 88% of the Sanctuary water exchanged during each tidal cycle (Day, 1981). The water area available to the aquatic biota was greatly decreased by the new harbour development and the species composition of the previous estuarine system was changed to become more marine in character (Day, 1981). The Ecoengineering to create a natural Sanctuary area therefore failed to achieve its aim, as did the engineering of the Mhlathuze River to create a sustainable and natural hydrodynamic regime as an alternative to the harbour.

Case-study 6 - Mangrove wetland creation: does it work?

While the above case-studies are Type A Ecoengineering, mostly modifying or controlling the hydro-physical regime through Ecohydrology, restoration by replanting, re-seeding or other species introductions is Type B Ecoengineering. While there are many examples, especially the very large studies of oyster and other biogenic reef creation or seagrass bed re-creation (e.g. Cerco and Noel, 2007; Katwijk et al., 2009; Wolanski and Elliott, 2015), here mangrove restoration is used as an example. Mangroves provide important wetland ecosystem services and societal benefits by filtering riverine sediment and nutrients, contributing to soil formation and helping to stabilize coastlines. They also provide shelter to the coast and absorb wave energy, thus protecting local communities against storm waves and high winds (Wolanski and Elliott, 2015). They further provide a habitat for marine organisms such as crabs and oysters, and a nursery for fish and shrimps which in turn support important artisanal and industrial fisheries. They support charismatic wildlife (e.g., the Bengal tiger) as well as producing organic litter (e.g., mangrove leaves) that supports both the local mangrove fauna (e.g., crabs) and, through outwelling, the estuarine and coastal pelagic food web. In addition, they provide timber, poles and wood fuel, fodder for animals and other economic opportunities through eco-based tourism (Hogarth, 2015; Barbier et al., 2008; Wolanski et al., 2009; Gedan et al., 2011; Wolanski and Elliott, 2015).

Despite all of this, mangroves are disappearing and indeed as early as 1975 the scientific community warned about the rapid loss of mangroves worldwide (Walsh et al., 1975; Duke et al., 2007). Hence, governments of developing countries are now increasingly recognising the value of mangroves and encouraging mangrove restoration efforts (e.g., Barbier et al., 2015). Early attempts to restore mangroves have consistently failed, leading to questioning as to why this is so and thus to renewed efforts to restore the biophysical conditions. This requires starting from restoring the original tidal hydrodynamics in order to try to mimic the natural conditions that may assist mangroves in colonising the site.

In essence, three types of mangrove management and restoration efforts have emerged. Firstly, there are long-lasting endeavours to sustainably exploit and manage large natural mangroves, e.g., the Matang Forest Reserve in Malaysia (Goessens et al., 2014) and the Sundarbans Forest Reserve in India (Ghosh, 2015). In the Sundarbans, the mangroves are recognised as protecting the human population from natural hazards and are managed as a wildlife area with a control of fisheries and active planting of endemic mangrove species on emerging mud banks. In Matang, the trees have been exploited for 100 years whereby they are cut, first by thinning and then by clear felling in blocks scattered amongst the forest. The replanting of seedlings, as well as natural recruitment by the tidal import of seedlings from the surrounding forest, both occur in the clear-felled blocks. Generally this strategy has been successful although over a century the forest has evolved towards becoming mostly mono-specific with *Rhizophora* because that species is preferentially planted in the clear-felled blocks (Goessens et al., 2014). The fish assemblages in the estuary still persist, despite many species being intensively harvested and probably over-exploited.

Secondly, there have been many attempts worldwide to plant mangroves in degraded or ill-suited areas (such as unsuitable tidal flushing conditions or soils) or without the active support and collaboration of the local communities. Such attempts have also failed from a variety of other causes, including excessive periods of immersion drowning seedlings, colonising oysters growing on the stem and toppling the seedlings, floating marine debris and macroalgae toppling or defoliating the seedlings, waves uprooting the seedlings, substratum erosion by waves, and seedling burial by migrating mud waves. There is also soil acidification by excavating shrimp ponds in mangrove soils, and people and livestock directly destroying the newly created mangrove forest without the community involvement needed to protect the restored mangroves (Wolanski et al., 2009; Primavera et al., 2011; Dale et al., 2014; Samson and Rollon, 2014; Wolanski and Elliott 2015; Gensac et al., 2015). Much of this mangrove restoration has been conducted without documenting the methodologies and the real costs of the work, and without adequate site assessment and remediation measures (e.g., to restore or generate suitable tidal hydraulics). In the few cases in the Philippines where such reforested and afforested mangroves have survived to form a forest after typically 20 years (longer data sets are practically non-existent), the canopy ratio did not differ significantly from that in natural forests but the stem density was much higher than that in natural forests (Table 6).

Thirdly, over the last 60 years there have been attempts to plant mangroves over sheltered, carbonate reef flats, i.e., where no such mangroves existed previously. These are judged successful because a forest has been created and may be the longest records of successful mangrove wetland creation. The successfully-created forest appears to have reached steady-state after 60 years (Asaeda et al., 2016) although the natural and planted forests differed significantly in terms of forest structure, density and species diversity, and the absence of the tree zonation characteristic of a natural system (e.g., Knight et al., 2008). The substratum of the planted forest had changed from sandy to muddy and crabs and fish were largely absent in the planted mangroves as tidal creeks were not created. It may therefore be argued that planting mangroves in areas where they previously were absent is ‘ecological gardening’ and not ecological restoration.

Discussion

The examples above show that there are many large and small Ecoengineering schemes and that a degree of restoration has been achieved in some cases whereas elsewhere several iterations are required even to achieve partial restoration, and often contrary to the designed objectives. However, artificial restoration is often Ecoengineering ‘gardening’, carrying out ecological modifications which are neither guaranteed to be necessary or successful but rather which make society (including ecologists) feel as though something is being done. The reality is that estuarine and coastal Ecoengineering has a poor track record and there has been a general failure of the science of restoration that aims for greater sustainability and resilience dependent on natural ecosystem processes (Hobbs, 2007). This engineering approach is, however, 'consistent with the Baconian-Cartesian-Newtonian philosophy of nature as reducible parts where the properties of behaviour of parts abstracted from nature can be maintained in order to provide human services' (Gattie et al., 2007). Often, the demand for restoration to provide instant human services circumvents basic principles of ecological engineering to produce a self-sustaining ecosystem (Mitsch and Jørgensen, 2007). Hence we recognize natural ecosystems as being self-designing, hierarchical in the context of a larger landscape and constituting a network, with Ecoengineering requiring a holistic eco-technological approach that integrates many, if not all, interacting parts and processes (Gattie et al., 2007).

Rationale of the approach and adequacy of the underlying science

As shown in the case studies here, Ecoengineering is important in coping with a historical legacy and centres on manipulating the ecosystem physics (Ecohydrology), hoping that the ecology then establishes. As shown by the mangrove examples, manipulating the ecology (e.g., by transplanting, restocking, Type B Ecoengineering) may be a last resort and only used when there are no natural propagules available to restock the system (the result of Type A Ecoengineering). Hence Type A and Type B Ecoengineering each has a defined purpose. However, it is important to separate engineering from Ecoengineering in that the former may or may not achieve a sustainable ecology, and usually damages the ecology, whereas Ecoengineering has this as its sole aim. The difficulty with Ecoengineering is that we often manage for one problem at a time and once that is solved then another problem appears. The sequence in dealing with the problems in an estuary may be important to the ultimate success of the programme, i.e., if you start with problem 2 (e.g. creating wetlands), then problem 1 (e.g. accommodating sea-level rise) becomes easier to deal with in the future. The danger is that the expediency often driving Ecoengineering may lead to omitting some serial steps in ecological processes that, in the long-term, may reduce the likelihood of attaining the later and desired successional stage. Instead of restoration to a more natural state, the examples here often attempted Ecoengineering for managing to ensure optimal social and ecological carrying capacity, to enhance assimilative capacity or to optimize particular ecosystem goods or services.

As estuarine scientists, we question whether Ecoengineering is easier or more difficult in dynamic systems such as an estuary. It may be more difficult due to the many drivers from both the inland (entire catchment) and marine environment that have a direct impact on estuaries. However, accommodating 'error' in estuarine Ecoengineering projects may be easier than in more 'sensitive' ecosystems such as coral reefs. We clearly know how to re-create intertidal habitats in low lying areas that were historically poldered by setting back flood defences and allowing the estuary to roll landward. Ecoengineering is relatively successful and rapid in this instance as long as natural Ecohydrology processes become fully operational. Attempts have been made to predict the type of intertidal habitats based on initial elevation and rate of sedimentation. However (as shown by the case studies here), the modelling may fail to accurately predict those, as well as habitat type and size – suggesting that we often do not know enough about the dynamics of these systems. Despite this, in other cases, the modelling is accurate and fits the empirical outcome.

The examples here also raise the question of whether we are asking too much of recreated/created sites, such as delivering different habitats and both ecological and social benefits, and whether we have sufficiently well-defined objectives and understanding. Most measures aimed at tackling the environmental consequences of human activities are to give society benefits (e.g. flood protection, lowered costs) or they are legally required (e.g. Barnard and Elliott, 2015). On balance, we probably do not know enough about the dynamics of heavily modified estuaries and how they may respond to large eco-engineering projects such as dyke realignment, especially with changing baselines. Hence, we can question, as shown by the examples here, whether we know enough about the dynamics to modify them. Perhaps, if there is one 'lesson learned' from estuarine restoration, it is that changing the estuary's structure changes its dynamics, but that the dynamics will often rearrange the structure and set a new equilibrium! Given the lack of data prior to industrialisation and urbanisation in most, if not all areas, it is difficult to predict the original base state and thus set realistic rehabilitation targets; it may even be the case that because of global change, original baselines may no longer be relevant and need to be revised (Duarte et al., 2009; Elliott et al., 2015; Kopf et al., 2015). Estuaries, by virtue of their widely varying spatial and temporal conditions, are very 'forgiving' systems, and thus have a high resilience (Elliott and Whitfield, 2011) that allows them to be 'fine-tuned' over time to achieve the desired results. Also, where unintended consequences occur, these can be used to good advantage, e.g., the creation of the largest and richest mangal in KwaZulu-Natal following the newly created Mhlathuze Estuary (southern half of Richards Bay).

Perkins et al. (2015) show that Ecoengineering can mitigate the effects of coastal development and show the successes, both through soft engineering to maximise habitat complexity, and hard engineering whose primary aim is to ensure human safety even if it is poor at protecting the ecology. Hence, if Ecoengineering can achieve the wins for ecology, economy and public safety through providing ecosystem services, then it gains acceptance.

The examples here, especially the Peel-Harvey, show the need to engineer both the water column and the substratum if the aim is to rehabilitate the whole ecosystem. However, whereas engineering the water column is relatively easy, and may involve removing any stressors (e.g. barriers) such that the ecology responds positively to the changed conditions, this may be less easy with the substratum. Ensuring good water quality would not necessarily ensure recovery of the sediments if, as is usual for fine sediments, they are a sink for trace metals; for example the Peel-Harvey shows that that sediment nutrient flux processes have now become decoupled from the influence of the overlying water column

(Kraal et al., 2013). However, leaving the contaminated sediments in place and allowing them to be capped with uncontaminated sediments may be a successful strategy (Simpson et al., 2002).

It is necessary to question whether we know enough about the systems being engineered, including what worked and what did not and, if it worked, why did it work and what timescale was required to determine if it worked. Sometimes rehabilitation takes years and maybe even decades (Borja et al., 2010; Duarte et al., 2015). However, more than many other systems, estuaries evolve constantly and so, provided rehabilitation change is moving in the right direction, this should be regarded as a positive outcome. The timescale will depend on habitats and processes that need to be created/restored and possibly the lifespan and turnover rate of the ecological components. Hence, firstly, restoration often needs to go through natural seral (successional) sequences to maximize the integrative processes that optimize the endpoint ecosystem. Secondly, we need to avoid expediency and be patient in allowing natural processes to develop naturally. Marshes develop naturally and optimally (perhaps with fewer invasive plants) if they are allowed to rebuild with natural suspended (clay, silt) accretion instead of deposition of coarse grained (sand) dredge material. Borja et al. (2010) and Duarte et al. (2015) suggested that the long turnover components such as higher vegetation take a longer time to recover than short-lived components. For example, mangroves take several decades whilst the timescale for functional estuarine intertidal habitats dominated by infaunal invertebrates is shorter (< 10 years).

As indicated above, some Ecoengineering (e.g. the managed realignment case-study) shows a poor understanding of the system, poor modelling and an inadequate understanding of accretion, developing a site differently from that predicted. Some compensation for loss elsewhere may be achieved, even if the habitats created are not the same as those lost. However, often there are either unclear, unattainable or no objectives for recreated sites. For example, initiatives aimed to create lagoon and mud (Paull Holme Strays) or mud (Chowder Ness) habitats showed that the former, which had less Ecoengineering preparation, was slower to recover but the latter recovered (by attaining the desired habitat) in a year, but then kept accreting until it ended up six years later as a high marsh area. This new system still supported wetland ecology but not what was required, hence a fundamental concern with wetland loss mitigation.

Furthermore, in restoration it is necessary to question whether the restored sites are where they should be or merely could be – as shown here, it is often the latter in that restored sites are where land was cheap/available and with no occupants rather than the best place for the ecology. Hence, for example, Ecoengineering may not deliver compensatory habitats such as a long-term mudflat if the new site is at the wrong tidal elevation. The design and location of the site will not deliver mudflats unless it has large open areas (without restricted water flows) and the appropriate elevation for creeks/channels to develop. However, creating intertidal habitats through realigning estuarine flood defences to offset future losses due to sea level rise is a good example of an Ecoengineering approach that can deliver an outcome in low lying areas without active management. This shows the importance of letting the Ecohydrology principles work over time, even a very long time, to allow the succession of habitats that will support the ecology.

This review shows that if we do not understand the physical system and the interactions among its structure and ecosystem processes, there is little chance of getting successful and sustainable ecological functioning. Such challenges cannot be met solely by hard engineering approaches designed to address symptoms (e.g. just raise the sea defences), but requires adaptive management aimed at addressing the causes of non-functioning habitats. Importantly, societal expectations of 'what is achievable', and thus the types of ecosystem services provided by our estuaries into the future, also need to be tempered in the light of these challenges.

Sequence and adequacy of Ecoengineering

Gray and Elliott (2009) and Wolanski and Elliott (2015) emphasised that, in most cases, measures to counter, reverse or compensate for human impacts are not engineering the ecology but rather just applying Ecohydrology and then let the ecology follow (*sensu* Type A Ecoengineering). However, here we extend this to show that as well as Ecoengineering the habitat, we can eco-engineer the ecology through restocking or replanting (Type B Ecoengineering) but often with outcomes less than what is wanted. Usually it is assumed that we have control over many aspects – such as the shape of an area, the flow rates and hence the currents, the type of vegetation, grazers and predators, and the topography/bathymetry (Gray and Elliott, 2009), much of which is related to water movement, quantity and quality. Despite this, many estuary rehabilitation efforts in semi-arid countries such as Australia and South Africa suffer from declining river flows entering estuaries and even a possibly simultaneous increase in nutrient/pollution loadings within the remaining volume of fresh water. It may be possible to improve water quality by better farming/industrial standards and pollution control but increasing human populations unfortunately require increasing amounts of fresh water from a finite river supply.

The case studies described here appear to confirm the adage that once we start modifying a system then we have to keep engineering it otherwise it reverts to some state we do not want, e.g., the St Lucia case, hardly the ecosystem sustainability we so often seek. Given that each estuary may respond differently to a particular Ecoengineering approach, we need to learn from responses and modify Ecoengineering activities accordingly – hence the need for adaptive management. One Ecoengineering activity may not automatically generate the desired result but rather there needs to be several or many feedback loops in any Ecoengineering approach. In some cases, this is a problem of technology – of the wrong sort, in the wrong place, of the wrong type, as shown, for example, by the Richards Bay and Humber (Chowder Ness) case studies. In the former example, the lack of success was not only a problem of technology (failure of engineered barriers) but because it was conceptually wrong – Richards Bay fundamentally could not have been converted from an estuarine lake into something completely different. One could argue that the technology (hard engineering) was very effective at creating a major export harbour, but the Ecoengineering approach was not an ecological success and never could be irrespective of the technology used.

Next we have to consider the adequacy of the spatial scale in Ecoengineering: is it the size of the engineering scheme or the size of the repercussions of the engineering that is important? Some relatively small Ecoengineering activities may have major repercussions for certain systems (so-called 'low hanging fruit', e.g., the flow regulation gates), whereas other rehabilitation efforts may require large Ecoengineering operations to achieve proportionally lesser goals. The main point is that Ecoengineering should aim to achieve an Ecohydrology equilibrium, i.e., a sustainable and more natural system. The Ecoengineering aims need to be defined so that failures of management actions can be identified and subjected to adaptive management. We emphasise that the Ecoengineering scheme should follow (and attain) the so-called 10-tenets and be ecologically sustainable, economically viable, technologically feasible, administratively achievable, legally permissible, socially desirable or tolerable, politically expedient, ethically defensible (morally correct), culturally inclusive and the effectively communicated (Barnard and Elliott, 2015).

Management considerations and governance obligations

A further question is who becomes responsible for the site following Ecoengineering. For example, the managed realignment sites discussed above usually were subject to management by the developer for five years after which the site may revert to a statutory body or even not be managed. Clearly, this is a failing as there will be no one responsible to remedy the site if it does not achieve the aims and so the site develops away from the desired status, as in the Chowder Ness case. Within such a timeframe, the system might even reach an equilibrium but in a state that is not desired; therefore there is the need to keep modifying it to keep it in a non-stable state. Hence we need to determine the restoration trajectories and assess whether a site is on a predictable trajectory even if it has not reached an equilibrium (Simenstad and Thom, 1996).

As emphasised here, we aim to get the best for nature and society but operators/regulators have to agree to act even if it does not achieve this endpoint – this could mean constant re-engineering (i.e., through adaptive management) such as the removal of accreted material to maintain a given topography. This requires engineering and technological measures (*a la* the DAPSI(W)R(M) framework) but also questions the competencies of the restoration/eco-engineering authorities and whether the nature conservation bodies have the engineering expertise and the engineers have the ecological expertise to achieve sustainable ecosystems. The evidence presented here suggests that this is rarely the case: engineers may only aim to increase structural biodiversity but that may not be sustainable and hence they need to increase and maintain functional biodiversity. Hence, all Ecoengineering efforts need a Management Committee consisting of policy-implementers, engineers, ecologists and conservationists.

As for the overall management aims of Ecoengineering schemes, perhaps there is a hierarchy in that flood or erosion protection for human safety is paramount, followed by the economy and finally ecology. Even within ecology, there is a hierarchy which may depend on the prevailing governance priorities. For example, a Ramsar area is focused mainly on bird ecology whereas a fish nursery area emphasises Ecoengineering for fishes. Furthermore, the ability and desire to undertake Ecoengineering relies on linking actions with the legal or policy obligations and having administrative bodies to fulfil the obligations, again reflecting the 10-tenets. The Ecoengineering needs linking to measures used to implement the Water Framework Directive and Marine Strategy Framework Directive in Europe, the Clean Waters and Oceans Acts in the United States and the corresponding governance elsewhere. Those Ecoengineering measures for different types of pressures require scientific Ecohydrology principles to determine what Ecoengineering should be done, how, why, where, and when.

Sustainability benefits

An Ecoengineering approach or measure needs to fulfil the 10-tenets criteria to be sustainable and often the economic tenet is the most important, hence the need to quantify the costs and benefits of the systems created. A port developer could take the view he knows the 'cost' of saltmarsh as he created some but this does not acknowledge the full 'value' such as the remaining ecosystem services. CBA (cost-benefit analysis) and CVM (contingent valuation method) are increasingly used to decide whether Ecoengineering worked or has at least achieved something for society (Luisetti et al., 2014).

Ecoengineering, again using the 10-tenets, has to be technologically feasible (i.e., can we do the engineering?) but the other nine tenets are required to get the Ecoengineering methods sanctioned or end-points accepted by society (Barnard and Elliott, 2015). Hence, the 10-tenets may need ranking to get successful Ecoengineering – technology and economics being most important, underpinned by the legislation and enforced by administrative bodies. We can question whether society is a primary driver for more Ecoengineering or whether nature conservation bodies just assume it to be. Conservation bodies should raise societal awareness regarding the value of societally and naturally healthy estuarine ecosystems, and then responsibly manage impacted systems.

The evidence here repeatedly shows that the aims of restoration projects and for Ecoengineering have been poorly defined, if at all, and managers have not determined what success looks like. For example, hard engineering for St Lucia failed (the Link Canal) whereas soft engineering (Back Channel through mangrove swamp) was relatively successful at getting Mfolozi River water into Lake St Lucia without associated high sediment loads. Success can therefore be demonstrated by achieving a concrete, quantified and pre-defined aim, suitable monitoring, adequate data, and an audit. Ecological success is often measured in the size and type of habitats re-created or restored (e.g., size of intertidal mudflat) against the targets (if any) set by the developers or nature conservation bodies. The functional capacity of created intertidal habitats to mimic the ecological roles of nearby natural sites is a better indication of success and is widely used to evaluate success in restoring saltmarshes (Mossman et al., 2012), invertebrate communities (Mazik et al., 2007, 2010) and estuarine bird communities (Atkinson et al., 2004, Mander et al., 2007).

Ecoengineering may aim for ecosystem compensation – to compensate the natural ecosystem for habitats (e.g., mudflat) or resources (e.g., fish stock) lost through development or where the user welfare has been affected (Wolanski and Elliott, 2015). Once the habitat or resource is restored then the users will eventually receive the benefits – if the scheme works – and compensation should result in environmental rehabilitation. As shown by the mangrove examples here, restocking and replanting are examples of Type B Ecoengineering rather than Type A in engineering the ecology as opposed to engineering the physical aspects. However, there are Ecohydrology feedbacks in that new mangroves, reedbeds and mussel beds will affect the hydrodynamic regime and if water functioning is decoupled from the sediment then this will not produce sustainable ecology. However, if both the sediment and the water characteristics are engineered to a satisfactory standard but there are no propagules for colonization then there will be no ecological change, hence the importance of restocking/replanting.

Concluding remarks

Ecoengineering approaches to restoration of estuarine/coastal systems involve both adding but also removing structures and impediments to natural ecosystem processes that are most likely to promote successful and sustainable ecology. The approaches cover many other large aspects not covered in the current review, such as pollution removal, such as via discharge controls, treatment and bioremediation, e.g., creating bivalve beds for nutrient stripping. As shown here, in some cases creating or re-creating the physical structure and habitat, thus allowing the ecology to recolonise (Type A Ecoengineering), may not be sufficient, and so Type B Ecoengineering (recovering ecology by restocking or replanting) will be required.

Both soft and hard engineering are valuable in different Ecoengineering schemes depending on the circumstances and available information. The overall aim is to achieve the 'triple wins' – for human safety, economy and the ecology and a soft engineering approach is preferable as a first option and will often be cheaper than a 'hard' approach.

The examples presented in this review emphasise the need for combining Ecoengineering with Ecohydrology in order to give a sound scientific base but also to have resonance with policy and management. Under the right conditions we can achieve success, we know what success looks like but also where and why the Ecoengineering approaches fail. We fully realise that a pristine, original state cannot be achieved because of human pressures but that we can achieve environmental benefits to maximise the remaining natural potential; however, baselines and target situations may have to be amended not least because of changing prevailing environmental conditions. Finally, we have shown that any created/re-created/restored system is unlikely to reach a natural equilibrium with appropriate spatial and temporal scales but that at least as scientists and environmental managers we are now aware of the limitations in the approaches.

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Figures

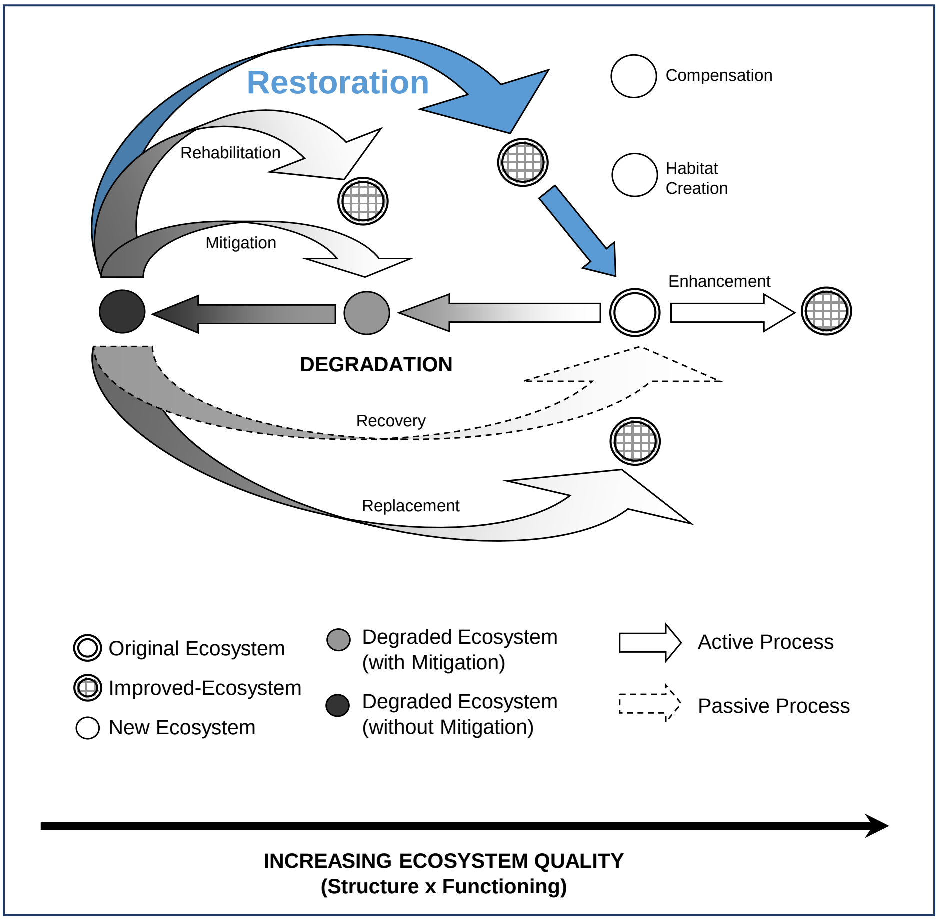


Fig. 1. Active and passive responses to ecosystem degradation (modified from Elliott et al., 2007).

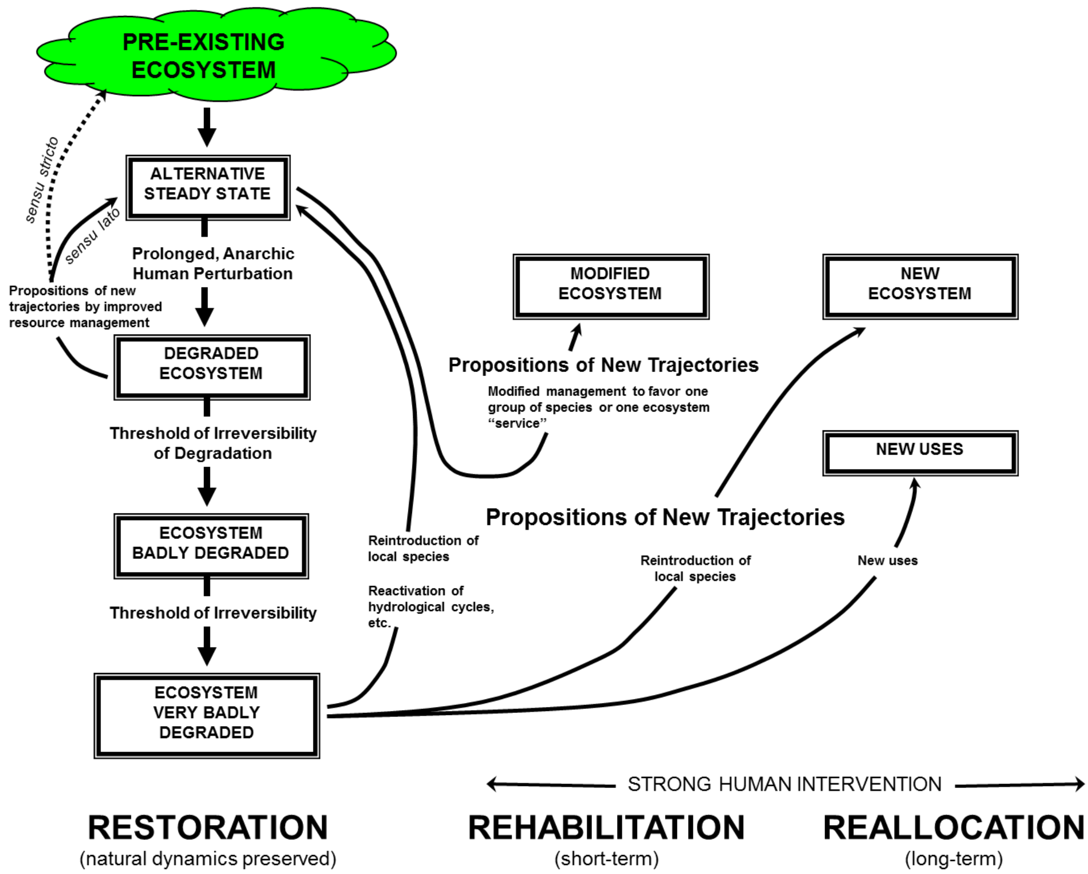
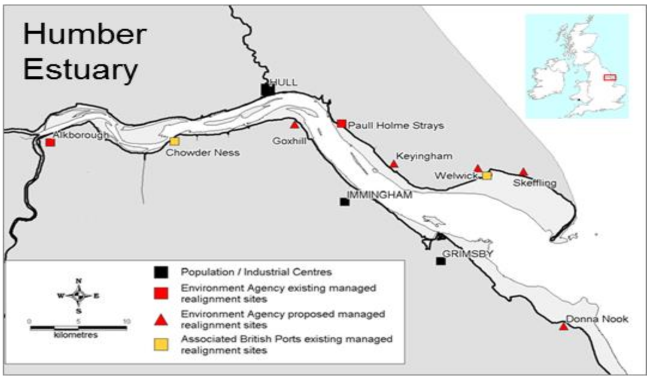


Fig. 2 Alternative ecosystem trajectories over 3 phases: restoration, reallocation and rehabilitation plus 'thresholds of irreversibility' to complete restoration to pre-existing conditions (from Aronson and Le Floc'h 1996).

Time		2003		2008		2013
Evolution of habitats	Arable land	Site creation	Shallow lagoon and mudflat →	Large mudflat and scattered saltmarsh →	Mudflat and saltmarsh →	Large saltmarsh and scattered mudflat
Sediment deposition & elevation	N/A		Rapid accretion at lowest elevation →	Steady increasing elevation		
Benthic macrofauna	N/A		Change from early colonising species to typically estuarine species →	Benthic communities present at low elevation typical of estuarine mud		
Evolution of foraging waterbird assemblage	N/A		Rapid colonisation by estuarine birds →	Dominance of benthivorous species ie. Waders and Shelduck (*Tadorna tadorna*)		

Fig. 3. Simplified conceptual diagram of the physical and ecological development at the Paull Holme Strays Realignment Site, Humber Estuary, UK (after Mander et al., 2007; Mazik et al., 2007, 2010; Brown 2014).

(a)


Humber Estuary
HULL
Alkborough
Chowder Ness
Goxhill
Paull Holme Strays
Keyingham
Welwick
Skeffling
IMMINGHAM
GRIMSBY
Donna Nook
Population / Industrial Centres
Environment Agency existing managed realignment sites
Environment Agency proposed managed realignment sites
Associated British Ports existing managed realignment sites
N
W
E
S
0
5
10
kilometres

(b)

(c)

(d)

Fig. 4. Map and Photographs of Chowder Ness, Humber Estuary, UK – (a) Map showing the Humber Estuary habitat recreation (managed realignment) sites; (b) pre-Ecoengineering 2006, (c) immediate afterwards 2007, and (d) long term 2015 (with the dyke walls grassed over) (Photographs M. Elliott)

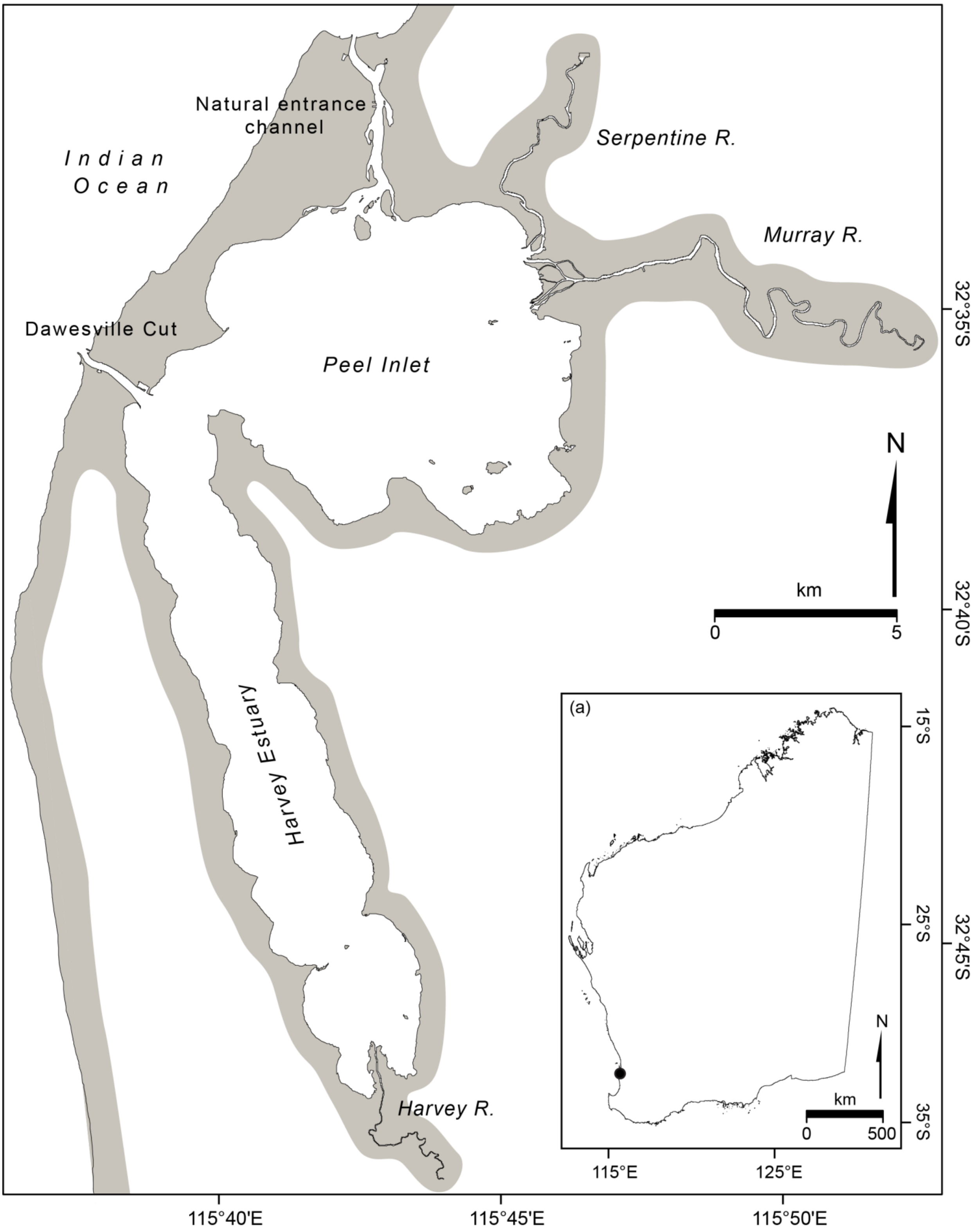


Fig. 5. Map of the Peel-Harvey Estuary, including its location in Western Australia (inset a).

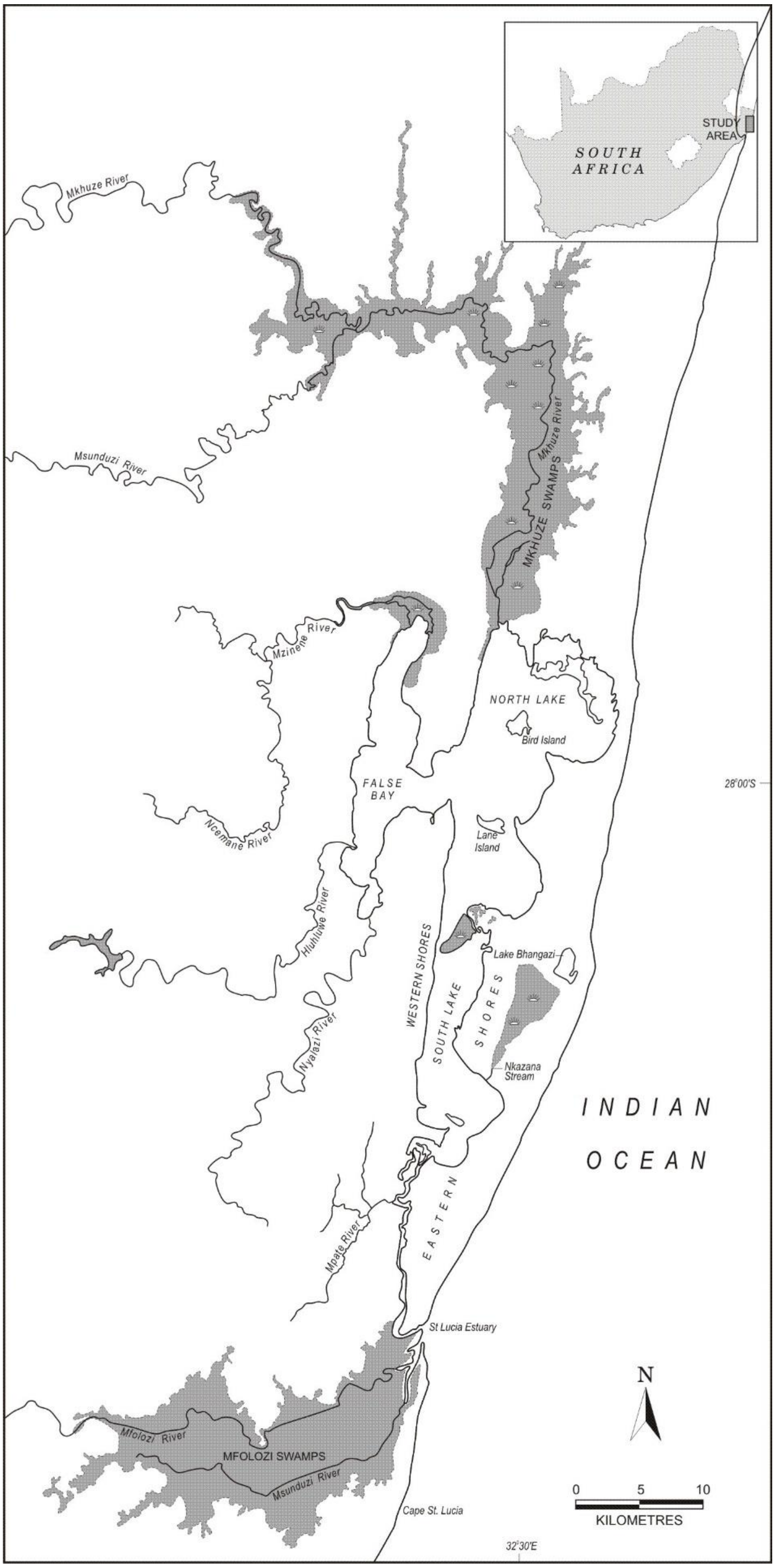


Fig. 6. Map of the St Lucia system, South Africa, showing places mentioned in the text.

Fig. 7. Aerial photograph of the St Lucia mouth showing the partially constructed groynes designed to stabilize the north and south banks of the estuary. In the mid-left of the picture is a stockpile of dolosse used to reinforce the groynes adjacent to the sea (Photo: R.H. Taylor, 1976).

Fig. 8. Aerial photograph showing the partially constructed intake works on the bank of the Mfolozi River in the foreground, with the Link Canal to the St Lucia Estuary under excavation in the background (Photo: R.H. Taylor, 1983).

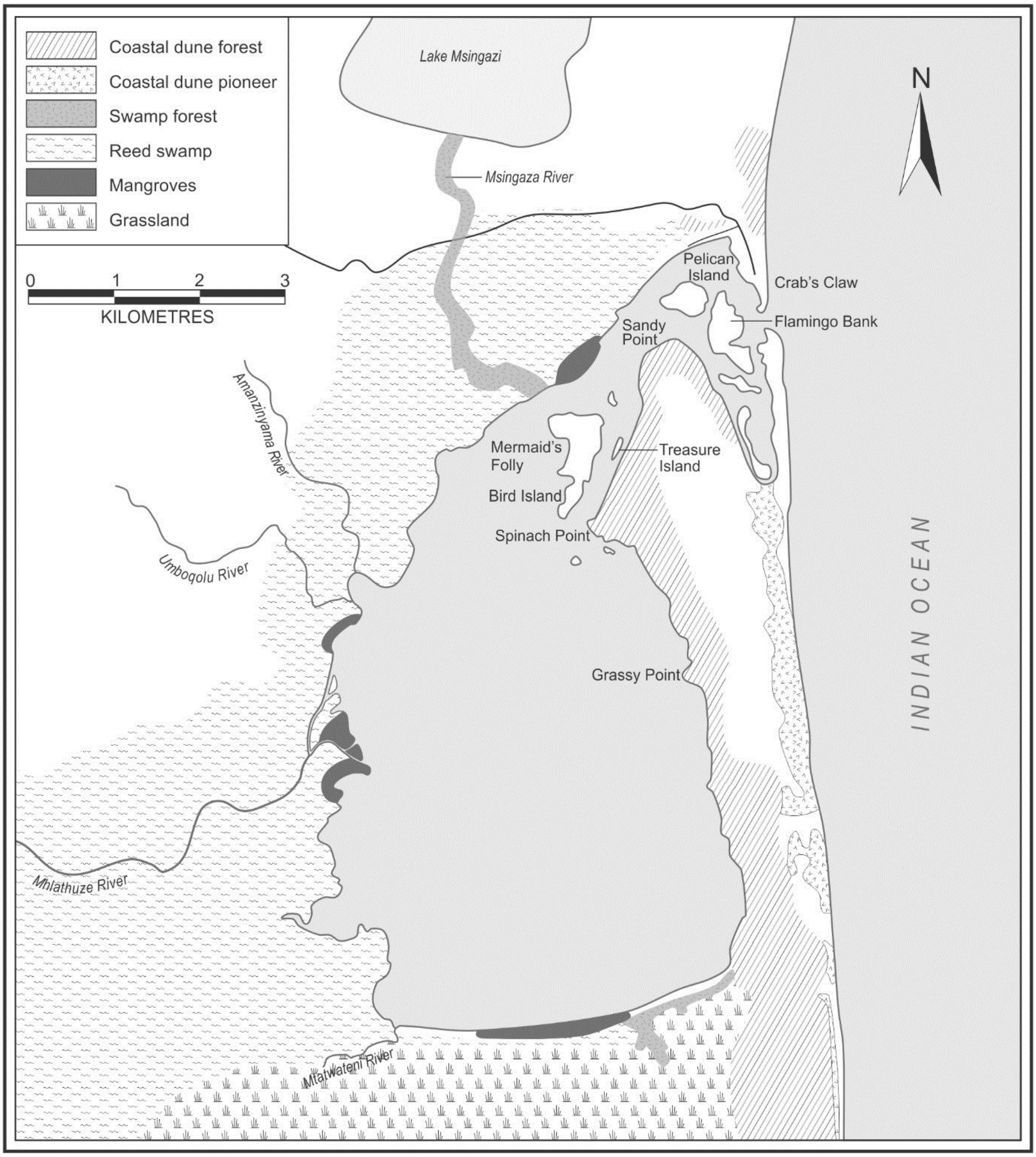


Fig. 9a. Map of Richards Bay, South Africa, in its natural state (1964).

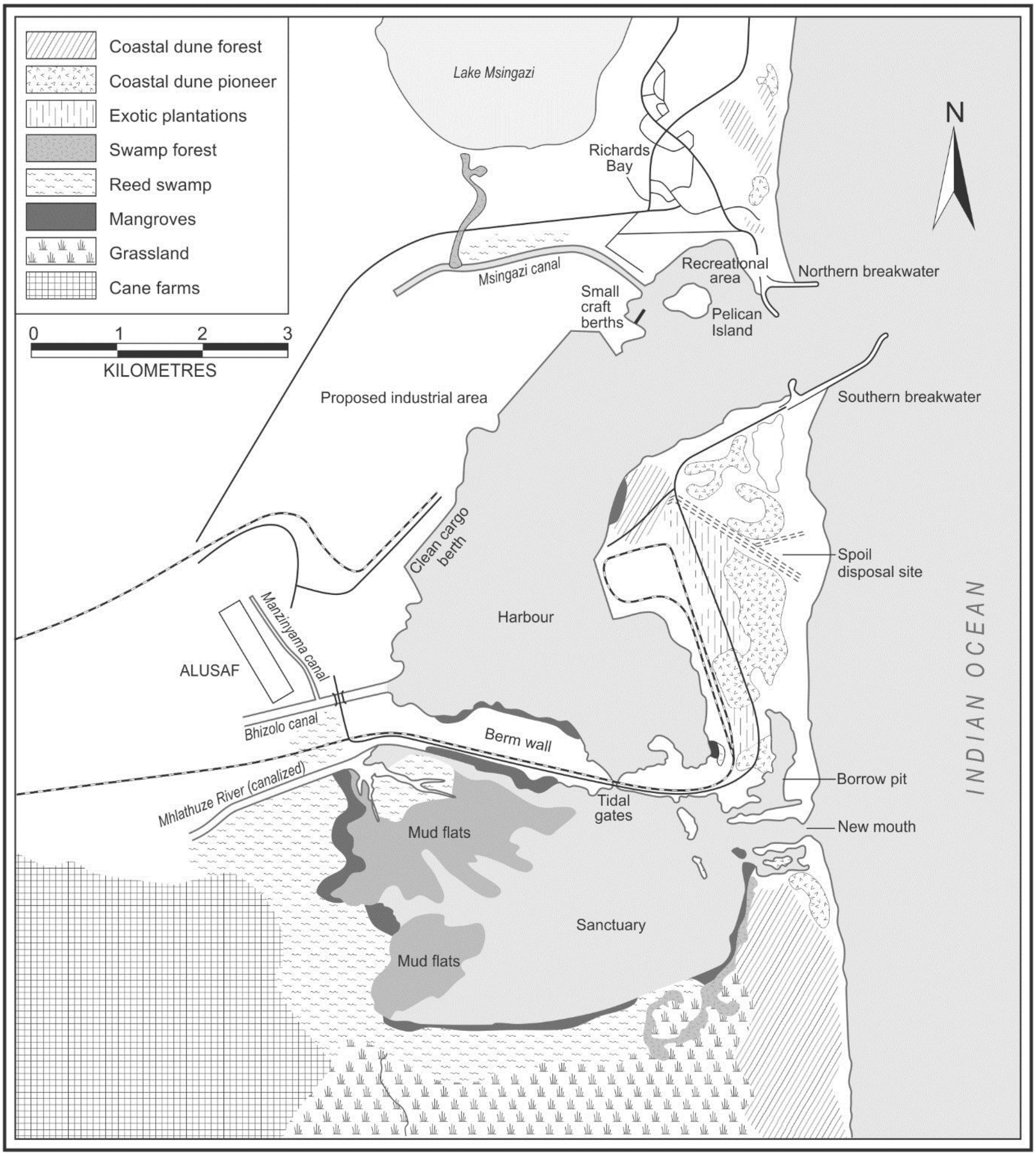


Fig. 9b. Map of Richards Bay, South Africa, area following the construction of the Richards Bay harbour in the north and creation of the Mhlathuze Estuary (Sanctuary) in the south (1976).

Table 1. Ecohydrological measure categories, types and examples (modified from Wolfstein et al, in press; van Wesenbeeck et al, 2014; REFORM wiki 2015)

Category	Ecohydrological measure type	Examples
Hydrology / Morphology	Measure to reduce tidal range, asymmetry and pumping effects and/or dissipate wave energy	Breakwater, dyke and levee, dune, seawall, storm surge barrier, groyne, managed realignment, beach nourishment and/or slope development, oyster reef, coral reef, artificial reef, macrophyte planting (e.g. saltmarsh, seagrass, mangroves)
	Other measures for flood protection	Flood barrier, floodway/diversion canal, retention pond, dam/weir
	Other measures to stabilise coasts or improve morphological conditions	Groyne, gabion or other rock armour, coastal vegetation, coastal terracing, revetment
	Measure to decrease the need for dredging	Controlling sediment erosion from catchments, improved management of deep-draught vessel entry (e.g. tidal phase timing), more stringent justification of channel width/depth requirements
	Zoning measures	Managed realignment, set back lines, building codes, multiple use zoning, special protection areas
	Measures to stop or reverse subsidence due to extraction of water and minerals	Aquifer recharge schemes; reinjection by oil and gas field production water
	Measure to restore longitudinal or lateral connectivity	Hydrological barrier removal (e.g. opening channels, removing weirs), wetland restoration, reducing water abstraction, increasing water release from dams for environmental flows
Physical / Chemical Quality	Measure to reduce nutrient loading (point and diffuse sources)	Removal/redesigning of land drainage systems (e.g. water sensitive urban design), changed agricultural and urban practices (e.g. fertiliser management), riparian buffer zones, set back lines, improved land-use planning
	Measure to reduce persistent pollutant loading (point and diffuse sources)	Removal/relocation of discharge drains, improved waste treatment prior to disposal, changed land-use/industrial practices, riparian buffer zones,
	Measure to improve oxygen conditions	Oxygenation plants/bubblers, reducing nutrient loads (see examples under ‘measure to reduce nutrient loading’), increasing hydrological flows (see examples under ‘measure to restore longitudinal or lateral connectivity), improving habitat conditions for bioturbators
	Measure to reduce physical loading (e.g. heat input by cooling water entries)	Diffusers, bafflers, heat sinks
	Measure to reduce sediment inputs and sediment loading	Shore stabilisation, changed land-use practices

Biology/ ecology	Measure to develop and/or protect specific habitats	Multiple use zoning, special protection areas, habitat restoration, managed realignment, maintaining environmental flows and connectivity (see examples under ‘measure to restore longitudinal or lateral connectivity’)
	Measure to develop and/or protect specific species	Wildlife corridors, harvest quotas, minimum size at capture, restocking, habitat protection (see examples under ‘measure to develop and/or protect specific habitats’)
	Measures to retain or restore natural gradients & processes, transition & connection	See examples under ‘measure to restore longitudinal or lateral connectivity’ and ‘measure to develop and/or protect specific habitats’
	Measure to prevent introduction of or to eradicate/ control against invasive species	Multiple use zoning, quarantine areas, vessel risk assessments, hull cleaning measures, biofouling, ballast water management, early detection pest monitoring programs, pest incursion response plans
	Measure for direct human benefit of ecological attributes	Multiple use zoning, special protection areas, habitat restoration, species protection
Human safety	Measure for early warning/evacuation of natural disasters	Real-time flood forecasting, improved predictive systems, evacuation procedures, storm shelters and refuges
	Measure for improved resilience of housing and industry	Coastal set-back/roll-back, improved land-use planning, improved building design, see other examples under ‘measure to reduce tidal range, asymmetry and pumping effects and/or dissipate wave energy’, ‘Other measures for flood protection’ and ‘Other measures to stabilise coasts or improve morphological conditions’

Table 2. Trade-offs and lessons learned from ecological engineering initiatives

Example	**Ecoengineering/ Ecohydrology Compromise**	**Lessons Learned**	**Literature**
'Fish friendly' tide gates regulating dyked surgeplain drainage	Regulate tidal inundation of channels and ditches behind dykes while allowing fish passage	TRADE-OFF: improve connectivity to tidal waters vs. deleterious effects of muted tide Very mixed results; depends on coastal setting and species of concern; some success for finfish and macroinvertebrates in Australia, but result in relatively poor connected or unsuitable, volitional habitat for estuarine-dependent anadromous salmon in north-west North America; general increase in hydrologic connectivity compared to traditional tide gates, but still result in degraded natural water quality, channel geomorphology and conditions for aquatic plants and invertebrates characteristic of natural tidal wetlands; require on-going care and maintenance; seldom rigorously evaluated or adaptively managed; compromise or attractive nuisance?	Giannico and Souder (2004, 2005); Beauchard et al. (2011); Boys et al. (2012); Glamore (2012); Greene et al. (2012); Reiner (2012); Franklin and Hodges (2015)
Artificial beach sediment nourishment	Artificially substitute beach sediment when along-shore source is deficient	TRADE-OFF: subsidize loss of natural sediment delivery to beaches versus lack of sustainable processes, expense and ecological impact; preferred alternative to engineered coastal protection structures; coastal erosion exacerbated by sea level rise; financial and economic benefits often outweighed by real costs to ecosystem goods and services; regular nourishments leave system in perpetual state of ecological disturbance; return period for repeated nourishment typically 5 yr	Speybroeck et al. (2006); Kabat et al. (2009); Schlacher et al. (2012); Manzanera et al. (2014); Martino and Amos (2015); de Vriend et al. (2015)
Installation of large wood	Anchor large wood in tidal wetland, channel and marine shoreline ecosystems to	TRADE-OFF: create more natural shoreline features and erosion protection vs. lack of natural dynamics; mimics large wood recruitment limitation; preferable to engineered shoreline 'hard' armouring; limited to no dynamics; no function in natural disturbance; seldom sustainable in high energy environments	Gonor et al. (1988); Everett and Ruiz 1993; Maser and Sedell 1994; Larson et al. (2001); Simenstad et al. (2003); Shields et al. (2004); Kail and Hering (2005); Piégay et al. 2005; Manners and Doyle 2008;

	mimic natural recruitment		Heathfield and Walker (2011); Kennedy and woods 2012; Rich et al. (2014)
Excavation of tidal channels	Excavate tidal channels to circumvent time required for natural processes of dendritic network formation to work	TRADE-OFFS: desire for instant gratification and jump-starting specific functions vs. restoration of natural pattern and rate of tidal channel geometry; engineering tidal channel networks in restoration design that often approaches 'gardening'; natural erosion of channels with renewed tidal inundation greatly dependent on relict imprint of original channels, substrate type and compaction and vegetative root mass; highly developed tidal geometry principles and hydrogeomorphic modeling provide sufficient understanding to predict drainage channel characteristics but not necessarily spatially explicit form; excavating simple 'starter' channels can provide basis for development of more complex, equilibrium network, but over-excavation can delay or sidetrack emergence of naturally complex network	Zeff (1999); Williams and Orr (2002); Williams et al. (2002); Hood (2002, 2004, 2006, 2007a&b, 2014); Fagherazzi and Sun (2004); Lawrence et al. (2004); D'Alpaos et al. 2005; Rozsa (2012)

Table 3. Summary of Ecoengineering and Ecohydrological events associated with the Humber realignment site schemes.

Site	Created	Size (ha)	Ecoengineering interventions	Ecohydrological characteristics	Ecological consequences
Paull Holme Strays	**2003**	**75**	**Creation of two breaches in the existing the line of defence and construction a new line of defence *approx*.500m inland.**	**Accretion of sediment due to tidal inundation - ranging 8.1cm at the highest elevation (3.15m Ordnance Datum) and 118cm at the lowest elevation (2.78m OD) over the first 10 years (Brown, 2014).**	**Rapid evolution of the site, from farmland habitats to estuarine mudflat and saltmarsh. Mudflats at lowest elevation supporting invertebrates and waterbird communities similar to those found on natural mudflat. However, saltmarsh coverage is increasing dramatically due to elevation gain.**
Alkborough Flats	**2006**	**370**	**20 m breach in line of defence + partial lowering of defences over 1500m to act as a weir and permit overtopping in extreme events. Half of the realignment site was set at 5.1m OD with the remainder of the site set at 5.4m OD.**	**Small breach control tidal inundation at the site.**	**Rapid evolution of the site, from farmland habitats to reedbed, saltmarsh and mudflat. Overall the site has enhanced the diversity of fish species found in that part of the estuary and appears to be acting as a nursery area for flounder. Invertebrate diversity and density is higher inside the site, compared to the impoverished**

					communities outside, but the assemblage is not typically estuarine.
Welwick	**2006**	**54**	**Existing seawall was removed over a length of 1,400m and new seawall was created inland. Two breaches were created in saltmarsh fronting the site. The land was re-profiled to increase the extent of lower areas where mudflat could develop**		**Extensive saltmarsh development and very small area of mudflats associated with the breach area.**
Chowder-ness	**2006**	**15**	New flood defences were created at the rear of the site. Although 200m remain, 570m of the existing seawall was removed in a series of stages to a level of approximately 1.6 - 2.0m OD.	**Following sediment accretion, site ceased to be inundated on neap tides after six years (Morris et al., 2013).**	

Table 4: Summary of some of the major engineering and Ecoengineering interventions and, where known, their Ecohydrological and ecological consequences, in the Peel-Harvey estuary and catchment from the early 1990s. Specific focus is given to the Dawesville Cut (*) built in 1994, and its Ecohydrological and ecological effects in the main basins of the estuary.

Time frame	**Engineering and Ecoengineering interventions**	**Ecohydrological characteristics**	**Ecological consequences**
Early 1900s	Desnagging of wetlands and rivers to increase flow rate of waterways and mitigate waterlogging of the land in winter. Channels then larger drains built to increase rate of water runoff [1; 2]	Reduced ability of floodplain wetlands to dampen river flows, leading to erosion and siltation issues in the estuary and wetlands [1; 2]	Unknown
1920s-30s	Further major drainage and river diversion systems successively built (e.g. Peel Main Drain, Serpentine River Diversion, Harvey Diversion Drain) in addition to straightening of river bends and further desnagging to reduce waterlogging of areas targeted for agricultural development [1; 2]	Further erosion and siltation issues in the estuary and wetlands, leading to additional land waterlogging problems and escalating cycle of drain building [1; 2]	Reduction in freshwater invertebrate, fish and mammal fauna noted [2]

1930s-early 90s	Further land drainage and clearing to accommodate agricultural boom [1; 2]. Eight dams built across various tributaries [5]	Further reduction in freshwater flows entering estuary, and loss of many seasonal wetlands. Significant nutrient pollution of the estuary via land drainage networks and other catchment runoff [2; 5]	Major macroalgal growths in Peel Inlet, particularly since 1960s (up to 60,000 t dry wt/y in 1979), and toxic blue-green algal blooms (*Nodularia spumigena*) in Harvey Estuary since late 1970s [2; 9; 18] Major loss of resident seagrass [10] Increased densities of various weed-associated fish species in Peel Inlet [7], but reduced fish densities and fish/crab kills near *N. spumigena* blooms [8; 14] Declines in Black Swan numbers [2]
1994*	Construction of a second entrance channel (Dawesville Cut) at the northern end of the Harvey Estuary to increase tidal flushing of the system and mitigate eutrophication and algal bloom issues.	Water exchange between estuary and ocean tripled [2]. Average salinity increased (20-30 pre-Cut to 35-40 post-Cut; [15]) 2-4 fold drop in TP and TN in water column [15] Little change in sedimentary TP, TN and organic content concentrations, though sites near Cut show some decline [6]. Estuary still has extensive monosulfidic black ooze deposits, anomalously high in iron monosulfides, and an excess of sedimentary organic matter [11; 12]	Major reduction in cyanobacterial blooms (none recorded 1999-2009; [15]) Macroalgae biomass far lower in Peel Inlet, but higher (as well as seagrass biomass) in Harvey Estuary [13; 18]. Loss of riparian plant species with greater freshwater/lower submergence needs, and increases in those with higher salt/inundation tolerance. Greater shore erosion negatively impacting some species [3] Major decline in benthic macroinvertebrates, i.e. mean density reduced to one third; mean species richness and diversity significantly lower; reduction in environmentally-sensitive crustacean species and increase in opportunistic polychaete species [17] Nearshore fish species richness increased, and while mean density declined just post-Cut (1996/97), increased in 2008-10 to levels as high or higher than pre-Cut. Assemblage increasingly dominated by marine species, and while prevalence of plant-associated species declined just post-Cut, has now increased to similar levels as the pre-Cut [16; 20] Crabs time their use of estuary differently and have an altered biology, i.e. 0+ cohort now enter the estuary

			earlier and grow faster in first months of life; females become ovigerous earlier and emigrate to sea earlier (4)

1-Bradby (1997); **2-**Brearly (2005); **3-**Calvert (2002); **4-**de Lestang et al. (2003); **5-**Environmental Protection Authority (2008); **6-**Hale and Paling (1999); **7-**Lenanton et al. (1984); **8-**Lenanton et al. (1985); **9-**McComb & Humphries (1992); **10-**McComb & Lukatelich (1995); **11; 12-**Morgan et al. (2012a, b); **13-**Pedretti et al. (2011); **14-**Potter et al. (1983); **15-**Ruibal-Conti (2014); **16-**Veale (2013); **17-**Wildsmith et al. (2009); **18-**Wilson et al. (1997); **19-**Wilson et al. (1999); **20-**Young & Potter (2003)

Table 5. Summary of Ecoengineering and Ecohydrological events during the manipulation of the Mfolozi/St Lucia estuarine system (Fig. 1) over the past 83 years.

Time frame	Ecoengineering interventions	Ecohydrological characteristics	Ecological consequences
1932	Artificial breaching of the beach berm at the mouth of the St Lucia Estuary to prevent flooding of sugar cane fields in the upper Mfolozi floodplain (water level = 4.3 m above mean sea level prior to breaching).	Massive outfow of sediment rich floodwaters for two weeks. Trees washed out to sea cause diversion of coastal shipping offshore.	Reduced scouring of the St Lucia Estuary 'bay' when compared to a natural breaching event
1933-1936	Wilson's Drain and Warner's Drain excavated along the Msunduzi and Mfolozi watercourses respectively (mainly to reduce inundation of floodplain sugar cane fields during river flooding).	Major reduction in the effectiveness of the Mfolozi floodplain swamp to filter sediments from the river water prior to entering the St Lucia Estuary.	Sediment accumulation in the St Lucia Estuary, especially during the closed mouth phase.
1952	New artificial mouth created for the Mfolozi River to the south of the St Lucia Estuary to reduce sediment input into the latter system.	Loss of Mfolozi River water to the St Lucia lake system but high sediment load waters, especially during floods, flow directly out to sea.	Extreme hyperhaline conditions and increased ecological stress in Lake St Lucia during prolonged droughts.

1953-1956	**Dredging of accumulated sediments in the St Lucia Estuary mouth region commences. A dyke is built to prevent the Mfolozi River from relinking with St Lucia Estuary.**	**The new Mfolozi Estuary and St Lucia Estuary mouths open and close independently of one another.**	**Ecological connectivity between the Mfolozi and St Lucia aquatic systems effectively ceases.**
1960-1968	**Two groynes, approximately 100 m apart, and comprising bags filled with a cement-sand mixture and dolosse (680 kg each), are constructed in the St Estuary mouth (Fig. 2).**	**This attempt to maintain a permanently open estuary mouth was partially successful but results in major ingress of seawater to Lake St Lucia during drought periods.**	**Extreme hyperhaline conditions and increased ecological stress in Lake St Lucia during prolonged droughts.**
1967-1969	**Dredging of the Narrows north of the St Lucia Estuary was undertaken to improve the connection between the St Lucia Estuary and Lake St Lucia.**	**Under drought conditions and an open estuary mouth, large volumes of seawater flow into Lake St Lucia and cause extreme salt loading of the system.**	**Extreme hyperhaline conditions and increased ecological stress in Lake St Lucia during prolonged droughts.**
1971	**Excavation of the 13.5 km long Van Niekerk's Canal through the Mkhuze Swamp in an attempt to bring freshwater directly to northern Lake St Lucia during a major drought.**	**Canal did not achieve the proposed goal of bringing freshwater to St Lucia and resulted in permanent loss of Mpempe Pan and draining of parts of the Mkhuze Swamp.**	**Functionality of the Mkhuze Swamp as an important feeder of freshwater into St Lucia was compromised by Van Niekerk's Canal.**
1975-1983	**Excavation of the 12 km long Mfolozi Link Canal with intake works (Fig. 3) and a sediment settling pond between the Mfolozi River and the St Lucia Estuary.**	**Works were commissioned in 1983 but shortly thereafter the construction was severely damaged by the Cyclone Demoina flood and never used.**	**Ecological connectivity between the Mfolozi and St Lucia systems would have been partially restored had the Link Canal become functional.**

1984	**Cyclone Demoina removes the St Lucia Estuary mouth groynes, dredger and severely damages the Link Canal system.**	**This Ecohydrological 'reset' of the St Lucia system improves the functioning of the entire system, except for the lack of an Mfolozi River connection.**	**Ecologically, the aftermath of Cyclone Demoina resulted in a healthy and productive St Lucia system for more than a decade.**
2002-2012	**St Lucia mouth closes naturally and is allowed to stay closed during an extended drought that lasts a decade.**	**In the absence of Mfolozi River flow into the system, more than 90% of the Lake St Lucia surface area is lost to evaporation.**	**Ecological devastation in North Lake and False Bay, with only remnant aquatic assemblages in South Lake, Narrows and Estuary.**
2012-Present	**Relinkage of the Mfolozi River with the St Lucia Estuary adopted as a new management strategy for the system.**	**Lake St Lucia salinities decline and the lake fills up with water.**	**Ecological recovery of Lake St Lucia currently in progress.**

Table 6. Mean and standard deviation of the canopy index[1] and stem density[2] of natural, reforested, and afforested mangrove forests at six locations in the Philippines. Recalculated from Samson and Rollon (2008).

Mangrove Habitat:		Mean	Standard deviation
Natural forest	Canopy index	2.6	0.9
	Stem density	33.9	10.7
Reforested	Canopy index	3.1	1.5
	Stem density	41	14
Afforested	Canopy index	2.3	0.7
	Stem density	71.4	21.3

[1] canopy index = total crown area/ area of substratum
[2] stem density = number of trees/100 m^2