**Direct and Indirect Effects of Estuarine Reclamation on Nutrient and Metal Fluxes in the Global Coastal Zone**

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**Abstract**

We demonstrate that land reclamation in estuaries is resulting in very large scale loss of intertidal area and disconnection of stored sediment with the water column. This process is not just causing loss of estuarine ecosystem services; it is also having a major deleterious impact on the ability of estuaries to retain nutrients and trace metals. The global scale of loss of estuarine wetlands and subtidal sediments has reached the point where the impact of this loss of estuarine retention is likely to be affecting coastal seas world-wide and possibly global element cycles.

**Introduction**

Fluxes of water and sediment through river systems are being greatly modified by human activities including agriculture, damming and engineering to reduce flood risk. (Syvitski and Kettner, 2011). In general, particulate sediment fluxes dominate the fluvial transport of all but the most soluble metals (Viers et al., 2009), and this is also true of the key biogeochemical elements C, N and P (Seitzinger et al., 2005), although the relative importance of dissolved nitrogen as a fraction of total nitrogen is increasing globally due to anthropogenic inputs (Seitzinger et al., 2010) particularly as nitrate (Jickells and Weston, 2011). These river inputs reach the coastal seas via estuaries which are important sites in the global biogeochemical system where riverine fluxes of dissolved and particulate matter can be modified. Decreasing flows and destabilisation of colloidal material due to changing salinities result in particle trapping within estuaries and continuing through the continental shelf (Dyer, 1997, Mantoura et al., 1991). In addition particle water interactions within estuaries can create exchanges between dissolved and particulate material modifying speciation and biogeochemical reactivity (Fitzsimons et al., 2011, Statham, 2012), processes that alter the cycling of sediments within an estuary.

Estuaries are transient environments on a geological timescale: on millennial timescales estuaries respond morphologically (under quasi-equilibrium) to processes such as sea-level change. In the coming century such equilibrium response will be tested as the rate of sea-level rise increases. (Statham, 2012, Andrews et al., 2011). However in addition to pressures from sea level rise, estuaries and near-shore environments are also subject to major changes resulting from the direct management of their shape and function by human engineering activities. The reclamation of the estuarine intertidal and subtidal seabed for habitation, industrial development, infrastructure and agriculture has massively changed the geography of estuaries. We have previously documented the loss of about 90% of the intertidal area of the Humber and Wash estuaries in the UK to reclamation in various forms over the last 3000 years, with most of the change over the last few hundred years. The loss by direct reclamation can be compounded by processes such as dredging in some systems (Turner, 1997). Based on detailed studies of 12 large estuarine systems in Europe and North America, Lotze et al. (2006) suggest overall a 67% loss of wetlands with an attendant massive loss of species diversity. This loss of wetlands predates the more recent increase in eutrophication pressures, and so reflects human civil engineering activity rather than eutrophication pressures.

The role of estuaries and salt marshes in modifying fluvial fluxes of metals and nutrients in estuaries has been a central theme of the research of Professor Tom Church, focussing particularly on the Delaware Estuary, an estuarine system which has also lost areas of wetlands, although there is now a programme of restoration in place (Kraft et al., 1992, Philipp, 2005). We have previously discussed the impact of estuarine land reclamation on nutrient budgets for some specific estuaries, particularly for nitrogen, for which the loss of intertidal area reduces the potential for denitrification (Andrews et al., 2006, Jickells et al., 2000, Jickells et al., 2014). Here in this volume to celebrate Tom Church’s career we consider the broader case that this reclamation may significantly affect the global estuarine flux of many biogeochemically important elements through the coastal zone.

**The scale of loss of estuarine habitat**

We have previously reconstructed the geography and sedimentation of the Wash and Humber estuaries, two of the main English river systems draining to the North Sea. In Table 1 we extend this approach to a larger group of North European estuaries, representing most of the river flow into the Southern North Sea, using a broadly similar, though less detailed, approach to that reported previously (Jickells et al., 2014, Jickells et al., 2000) and described in detail elsewhere (Parkes, 2003). This Table clearly shows that the reclamation described for the Humber and Wash is only an example of a process that has been carried out along much of the low lying north European coast. This large scale loss of intertidal area is accompanied by an attendant loss of important ecological habitat, loss of carbon storage which could in part offset atmospheric increases in CO2 and associated increased costs of man-made sea defences, representing overall a net loss of ecosystem services which can be financially valued (Luisetti et al., 2014, Lotze et al., 2006, Turner and Schaafsma, 2015). North east Europe is not unique in such reclamation with half of the estuarine wetlands of the Delaware estuary lost between 1780 and 1980 (Kraft et al., 1992), extensive losses in the Northern Gulf of Mexico (Turner, 1997) and extensive reclamation programmes currently underway in East Asia (Yang et al., 2011). Current global loss rates of coastal ecosystems are estimated at 1-5% per year (Duarte et al., 2008) and future sea-level rise may exacerbate this trend without a change in coastal management practices (Nicholls et al., 1999). In China 13380 km2 has been reclaimed since 1950 amounting to a 50% loss of coastal wetlands and 70% loss of mangroves, with further reclamation planned in coming decades (Wang et al., 2014). This reclamation includes large areas of major estuarine systems such as that of the Yangtze (Changjiang) river. Similar large scale loss of estuarine area has taken place in the Mississippi (Turner and Rabalais, 2003, Mitsch et al., 2001). In the case of both the Mississippi and the Yangtze (Chiangjiang), and elsewhere, the changes within the estuary are only part of a large scale pattern of engineered change within the catchment including damming and embankment (Mitsch et al., 2001, Turner and Rabalais, 2003). The Three Gorges Dam project on the Yangtze (Chiangjiang) is a graphic example, which is causing reduced sediment delivery to the coastal zone affecting the stability of the delta (Dai et al., 2011, De Vriend et al., 2011).

The scale and breadth of estuarine habitat loss to reclamation now amounts to a global scale engineering activity which started a few thousand years ago but accelerated a few hundred years ago, continues today and is likely to continue into the future, exacerbated by climate change and sea-level rise (Lotze et al., 2006). This, we suggest here, has not only major impacts on shelf seas adjacent to the river systems, for example, hypoxia evident off both the Mississippi and Yangtze (Chiangjiang) (Zhu et al., 2011, Rabalais et al., 2009) but potentially also global scale biogeochemical implications. In the following discussion we consider two related types of impacts of estuarine land reclamation: (1) the direct effect from loss of intertidal/subtidal sediments within the estuary leading to reduced estuarine denitrification and sediment stored; and (2) the effect of reclamation on estuarine suspended solid concentrations which in turn affect chemical reactions within the estuarine water column. These two effects are linked and affect all chemical fluxes to varying degrees, although the relative importance of the two impacts is not the same for different elements and species.

|  |  |  |
| --- | --- | --- |
| Estuary | Present Area km2 | Area 3000yr BP km2 |
| Humber | 111 | 1490 |
| Wash | 295 | 2566 |
| Thames | 24 | 650 |
| Scheldt | 134 | 5600 |
| Rhine | 1420 | 20300 |
| Ems | 918 | 10900 |
| Weser | 1130 | 8400 |
| Elbe | 2180 | 8250 |

|  |  |  |
| --- | --- | --- |
| Estuary | Current C storage 103t a-1 | 3000yrBP C storage 103t a-1 |
| Humber | 2.5 | 151 |
| Wash | 11.3 | 266 |
| Thames | 0.5 | 75 |
| Scheldt | 3.3 | 784 |
| Rhine | 32 | 2742 |
| Ems | 20 | 1002 |
| Weser | 26 | 620 |
| Elbe | 51 | 606 |

Table 1 Loss of habitat and carbon storage over the last 3000 years to reclamation in north east European estuaries (Parkes, 2003).

**Direct Biogeochemical Impacts of loss of estuarine sedimentation**

*Nitrogen*

The loss of intertidal sediments can alter the nutrient retention capacity of estuaries, such as when the loss of intertidal area reduces the capacity of the estuarine and coastal system for denitrification, leading to less nitrate retention and greater nitrate inputs to coastal zones (Jickells et al., 2014). We previously estimated an average denitrification rate for the sediments of the Inner and Outer Great Ouse and Humber estuaries as 0.64, 3 and 0.6 molNm-2yr-1 respectively (Jickells et al., 2014, Jickells et al., 2000). Similar rates have been reported for the Elbe (Deek et al., 2013), Yangtze (Wang et al., 2007) and Mississippi (Mitsch et al., 2001). In their very extensive review of denitrification Seitzinger et al., (2006) estimated a similar global average estuarine denitrification rate of 0.4 molNm-2yr-1 , and that globally about 25% of nitrogen inputs to rivers, estuaries and shelf systems are denitrified. Denitrification rates vary with many environmental factors including temperature, organic carbon, oxygen and nitrate concentrations (Seitzinger et al., 2006), but self-evidently any loss of intertidal sediment area will reduce the potential for denitrification. In addition loss of intertidal area will decrease water residence time within an estuary and therefore also decrease the potential for denitrification. Thus the loss of intertidal area must lead to a reduction of denitrification within estuaries and hence increased export of nitrate to coastal zones. To illustrate the possible scale of this effect, in Table 2 we have estimated the potential nitrogen retention by the European estuaries in Table 1 with their current areal geography and with their inferred area 3000 years ago, assuming a global average estuarine denitrification rate of 0.4 mol m-2yr-1 (Seitzinger et al., 2006), which clearly will vary as noted above. The results for the Wash, Humber and Elbe are broadly consistent with more comprehensive contemporary budgets for these systems (Jickells et al., 2014, Jickells et al., 2000, Deek et al., 2013). The key point from Table 3 is that the nitrogen cycle is being altered substantially in coastal seas as a result of both increased riverine inputs and also due to the loss of estuarine nitrogen retention (see also Mitsch et al., 2001, Deek et al., 2013). The effects of land reclamation are now at a scale that is significant in the context of historic rises in nutrient riverine inputs to these areas and the scales of reductions envisaged by current legislation; in the case of the North Sea for example the policy aim is for a 50% reduction in inputs of nitrogen and phosphorus (Claussen et al., 2009).

Table 2 Estimated percentage estuarine N retention of present fluvial N inputs for estuaries in Table1 assuming a denitrification rate of 0.4 mol m-2yr-1 and the areas in Table 1 for the present day and 3000 years ago. River N inputs are for the present day are based on OSPAR data for full details see Parkes (2003).

|  |  |  |
| --- | --- | --- |
| Estuary | Modern % N retention | % N retention with 3000 year old geography |
| Humber | 1% | 14% |
| Wash | 9% | 76% |
| Thames | 0.4% | 12% |
| Scheldt | 1.5% | 63% |
| Rhine | 3% | 44% |
| Ems | 17.6% | 209% |
| Weser | 10% | 72% |
| Elbe | 9% | 33% |

*Carbon*

In some ways nitrate is a special case, because denitrification is a bacterial process capable of nutrient retention and yielding gaseous products which are lost from the system. We now consider the more general case of particle water interactions and burial and the extent to which these are modified by loss of estuarine area.

A key issue here is the change in sediment burial. Estuaries do not represent the only site of sediment storage in coastal seas, there is extensive storage in other coastal wetlands (saltmarshes and mangroves), deltas and in some depocentres on the shelf or the shelf break. However, estuaries are important sites of sedimentation. In the North Sea for example 15-23% of sedimentation may occur in estuaries including the large estuarine embayments of the Wadden Sea and Wash (Eisma, 2009). There is rather little sedimentation across much of the North Sea outside of the Skaggerak/Norwegian trench region (Eisma, 2009). A similar situation has been suggested for the US East Coast, with most modern sedimentation taking place in the estuaries (Milliman et al., 1972), and also in the Yangtze (Changjiang) (Milliman et al., 1985) and Mississippi (Galler and Allison, 2008) systems. Thus the storage of particulate matter in the estuary is reduced by reclamation of intertidal and subtidal area which will affect the resulting input of fluvial particulate matter to the shelf and the storage of fluvial material within the estuary.

As illustrated in Table 1 for estuaries draining into the southern North Sea, the carbon storage in estuaries is massively reduced by estuarine land reclamation. Moreover the ‘pipe-like’ geography of engineered estuaries precludes particle storage resulting in ‘by-pass’ transport toward shelf-sea depocentres with at least some of this organic carbon degraded during transport with attendant release of CO2 and nutrients bound to the organic matter. The transfer of organic carbon from land to sea is a major global flux (Galy et al., 2015) and its efficiency as a CO2 sink depends on the effective burial of this carbon before its respiration (Hilton et al., 2015). Hence the loss of estuarine carbon storage has the potential to have global scale impacts on coastal seas, ocean biogeochemistry and on climate (Mcleod et al., 2011).

*Trace Metals*

Loss of estuarine wetland will also alter the cycling of river-derived sediments within the coastal zone, in particular whether they are trapped within the estuary or ‘by-pass’ it to the open waters of the continental shelf. This fluvial particulate matter is not geochemically inert even after burial. For example net dissolution of strontium, neodymium and calcium (Oelkers et al., 2012, Gislason et al., 2006, Lacan and Jeandel, 2005, Jeandel and Oelkers, 2015) from fluvial particles has a small, but geochemically significant effect on global budgets and isotopic balance. , The timescales of these dissolution reactions are relatively long, measured in the laboratory on scales of 100 days, and so may be sensitive to the sedimentary environment of dissolution, both in terms of redox and also the subsequent migration of solutes formed to the water column. Hence these processes may be sensitive to the loss of estuarine wetland sedimentation which changes the locus of river-derived sediment deposition to open shelf waters.

For calcium, strontium and probably neodymium, recent changes in sedimentation caused by engineering in estuaries are unlikely to have yet had a significant impact on the open ocean biogeochemistry of these elements, given their long ocean residence times. However, the more dramatic effect of sea-level lowering during glacials which, during the last glacial maximum eliminated the contempory continental shelf, may have been much more important, as has been argued for nitrogen (Galbraith et al., 2013). Jeandel and Oelkers (2015) further suggest that dissolution of fluvial sediments may also play a role in the budgets of transition metals including Fe, Mo, Cd, Cu and Ni and of Si (Oelkers et al., 2011). These trace metals which, may be deleterious to coastal ecosystems under certain conditions, but can act as nutrients under others, all have relatively short coastal water residence times and complex estuarine biogeochemistry which we discuss later (e.g. Church, 1986, Millward, 1995). In the case of iron at least, the ocean residence times are short enough that changes in estuarine storage capacity may have a significant effect on coastal systems and possibly beyond (see below) and this may also be the case for Cd, Cu and Ni, although complicated for these metals by the changes in inputs associated with anthropogenic activity.

Iron is a key nutrient that limits productivity in large areas of the open ocean (Boyd and Ellwood, 2010). The largest input of iron to the oceans is as particulate iron. This flux is more than 10 times larger than the inputs from other sources, but this input along with dissolved fluvial iron is generally assumed to be effectively trapped in estuaries and near shore systems (Jickells et al., 2005). Dissolution of even 1% of fluvial particulate iron (which is at the lower end of the dissolution seen for some other crustally bound elements Jeandel and Oelkers, 2015) would represent a significant source, equivalent to about 50% of the currently estimated atmospheric iron input to the oceans (Jickells et al., 2005). This estuarine release would require sufficient strong siderophore (iron binding) ligands to stabilise dissolved iron in solution (Boyd and Ellwood, 2010) and allow export to coastal waters (Hopwood et al., 2015b) where iron may in some cases be an important nutrient (Lohan and Bruland, 2008). Thus although a large amount of fluvial iron may be removed within estuaries, a biogeochemically important amount may still escape to the shelf seas (Hopwood et al., 2015b) where it may be removed to and recycled from sediments several times (Lohan and Bruland, 2008) leading potentially to export from the shelf to the ocean as suggested for glacial iron sources (Hopwood et al., 2015a). An inverse correlation between dissolved iron and salinity has been reported in the waters of both the English Channel and Arctic Shelf (Ussher et al., 2007, Klunder et al., 2012), consistent with such a fluvial source, including from particle dissolution, which is therefore at least reaching the open continental shelf waters in these areas, and potentially the open ocean itself.

Estuarine saltmarshes are also important sites for the processing of trace metals. Tom Church was one of the first to identify the importance of saltmarshes as a globally significant sink for uranium (Church et al., 1996). This has now been extended to suggest they are the dominant uranium sink (Windom et al., 2000) and can also affect the fluxes of radium and barium through estuaries (Moore and Shaw, 2008). As Church and his colleagues have also shown, different salt marshes with different sedimentary geochemistry may have a rather different impact on water column geochemistry (Velde et al., 2003), but what is clear is that the removal of the saltmarshes has the potential to greatly alter the biogeochemical function of the estuaries. Barium and uranium have ocean residence times of many thousands of years (<http://www.mbari.org/chemsensor/summary.html>) and hence any impacts of loss of salt marshes over recent decades will not have any short term impact on open ocean concentrations, but will have impacts on estuarine and shelf sea concentrations.

**Indirect Effects of estuarine reclamation**

*Suspended sediment matter*

Suspended sediment matter (spm) concentrations in estuaries are highly variable in space and time. Many larger estuaries contain a turbidity maximum zone (TMZ) composed primarily of resuspended material. The suspended sediment regime can be predicted from the mean tidal range and length of the estuary, increasing exponentially with both (Uncles et al., 2002). The loss of intertidal sedimentation area has important effects on estuarine suspended sediment regimes because it affects estuarine water flow patterns by enhancing flood tide dominance, which traps particles within the estuary, and also removes the capacity to then store this sediment leaving it available for resuspension. These effects can be compounded by artificial deepening of channels for navigation. The effect of these changes can be that reclamation can move an estuary to having higher spm, and potentially, through a regime shift or tipping point, to become hyper-turbid with suspended sediment concentrations of several 10s of g l-1 (Winterwerp and Wang, 2013). The theory of this evolution has been considered and validated in a series of North European estuaries including the Elbe, Ems, Loire and Scheldt (Winterwerp et al., 2013, Winterwerp and Wang, 2013, van Maren et al., 2015). The change to this hyper-turbid state can occur over decades as the stock of potentially resuspendable sediment builds up in an estuary after reclamation. Thus many estuaries are probably still responding to reclamation and subsequent changes, such as navigation dredging, that have taken place over the last decades and in many cases continuing to the present day. The biogeochemical response to future changes such as managed realignment and re-creation of intertidal sedimentation may therefore also take decades to develop, not least because the induced changes to the tidal prism may have unexpected and possibly non-linear effects on sedimentation over time (Winterwerp et al., 2013).

The development of high spm levels will clearly impact on estuarine algal blooms by decreasing light levels, although productivity under turbid conditions is still possible, albeit at low levels (e.g. Fichez et al., 1992). In addition particle water interactions affect the flux of fluvial material through estuaries which depend on spm concentations, the distribution between dissolved and particulate phases and also their residence times (Church, 1986, Millward, 1995), and so will be modified by reclamation as we discuss next.

*Phosphorus, cadmium and zinc*

The distribution of many elements in natural waters can be described in terms of a pseudo equilibrium between particulate (P) and dissolved (D) species of the form (e.g. Turner and Millward, 2002)

Kd = P µmol/kg sed/

D µmol/l water

Prastka et al (1998) and Ng et al (1996) have explored the implications of this kind of relationship for the transport of P, Cd and Zn respectively within estuaries. The general properties of a Kd model can result in four features of relevance to the discussion here (Turner and Millward, 2002, Millward, 1995, Tappin, 2002).

1. Loss of fluvial components from the dissolved phase to the particulate phase at low salinities due to mixing of high concentrations of sediment of riverine and offshore origin which are not fully saturated with the target element,
2. Buffering of dissolved concentrations (i.e. concentrations do not vary with salinity) by dissolved/particulate exchange within the estuary, so long as there is an adequate supply of particulate matter and adsorbed element of interest,
3. A decrease in dissolved concentrations beyond the estuary as suspended sediment concentrations fall moving into open coastal waters due to dilution with low dissolved and particulate concentration coastal seawater,
4. “Particle concentration dependent partitioning” effects (Turner and Millward, 2002).

These effects can be compounded in the case of these three elements at least by systematic changes of Kd with salinity reflecting the formation of dissolved Cd and Zn chloro complexes and sulphate competition for particulate P adsorption sites (Ng et al., 1996, Prastka et al., 1998) and additional effects of salt on organic/particle interactions (Turner and Millward, 2002). However, it should be noted that this kind of Kd formulation assumes fully reversible adsorption/desorption reactions which may not always be the case (Millward and Liu, 2003, Turner and Millward, 2002).

This impact of spm on the transport of fluvial inputs through estuaries may be particularly evident in turbid estuaries such as the Humber, Ems and Yangtze (Chiangjang) with a very large range of spm concentrations from >1000mg l-1 in the TMZ to <10mg l-1 in the coastal waters beyond the confined estuary. Estuarine management affects these particle water interaction processes in two ways. Firstly by increasing spm concentrations, reclamation can increase processes that transform dissolved species to particulate at low salinity and sustain estuarine buffering of dissolved P concentrations (Statham, 2012, Ruttenberg, 2003) as is seen, for example, in the Humber (Sanders et al., 1997), Yagtze (Changjiang) (Zhang et al., 2007) and Scheldt (van der Zee et al., 2007) . Secondly, in the absence of sediment storage due to land reclamation, the particulate matter within the estuary will eventually exit the estuary and mix with lower concentration offshore waters, when the Kd model then predicts that adsorbed element will be released. Detection of this effect is inherently very difficult using standard property salinity plots which are much more effective at discriminating changes at low salinity (e.g. Sanders et al., 1997). The key point is that the apparent effective retention of the element at low salinity is not fully realised.

This point can be illustrated by a specific example for particulate phosphorus (ParticP) and dissolved inorganic phosphorus (DIP). Prastka et al (1998) report a Kd (Partic P µm/kg/DIP um/l) value for the Humber of 24,000 l kg-1 for inorganic P and 30,000 for total P, comparable to values for the Amazon (42000), Zaire (38000) and Tay (77500) with rather lower values for the sandy sediments of the Wash (2500). The suspended sediment concentrations of total P in the Humber average 37.5 µmol/g with about 66% of this inorganic values, very similar to other estuaries (Prastka and Malcolm, 1994) and consistent with more recent data from the Yangtze (Changjiang) (Hou et al., 2009) Taking a value of 25µmol/g inorganic P and a Kd of 24,000 yields an estuarine DIP concentration, we can predict the buffered estuarine DIP concentration.

DIP = Partic P (µmol kg-1) /Kd

DIP = 37.5x103 / 24,000 = 1.6µM

which is broadly consistent with the observed concentrations of DIP throughout the lower reaches of the Humber (Sanders et al., 1997), and Yangtze (Changjiang) (Zhang et al., 2007) and somewhat lower than seen in Scheldt (van der Zee et al., 2007).

Now consider if this particulate material is advected into the North Sea where it is diluted by low DIP coastal seawater with low DIP reflecting lower oceanic concentrations and active biological DIP uptake. Suspended sediment concentrations in the North Sea are of the order of <1-5mg/l with at least some of this material phytoplankton debris. DIP concentrations in winter and summer are 0.8 and </= 0.1µM respectively (Radach and Lenhart, 1995).

Upon the advection of spm with 37.5 µM/g into North Sea waters with DIP concentrations of 0.1 (summer) or 0.8µM (winter) DIP (e.g. Radach and Lenhart, 1995), then based on Kd, partic P will desorb and the maximum amount that can be desorbed is of the order of 0.1µM/l equivalent. Hence in neither case is the pool of particulate P sufficient to sustain the Kd predicted concentration of 1.6µM DIP, and effectively all of the DIP adsorbed at low salinity will be desorbed, assuming the Kd distribution is reversible.

Ng et al (1996) employ a sophisticated 2D model for the Humber simulating water and sediment transport and apply a varying Kd value along the estuary to best fit the data, consistent with competition for sediment adsorption sites by seawater cations as salinity increases. As seen above for phosphorus, their Kd model predicts desorption of Cd and Zn from estuarine spm as it is advected seaward. This is an inevitable property of a Kd formulation, but whether the effect is significant depends on the actual Kd value, and clearly at least for these elements the effect is significant. In the case of P, Cd and Zn it appears that the observed Kd and ambient concentrations are such that this effect is very pronounced and hence that adsorption at low salinity will be reversed if the sediments are not effectively sedimented within the estuary. Such desorption may also be important for other metals and organic contaminants as well (Turner and Millward, 2002). Hence reclamation of estuarine intertidal and subtidal sediments has the potential to alter the cycling of these elements at least within estuarine waters and generally to decrease retention in estuaries and allow key biogeochemical species such as nutrients to reach coastal waters.

**Conclusions**

Globally it has been suggested that estuaries retain 40% of fluvial total N inputs and 66% of total P inputs to the ocean (Tappin, 2002). Managed realignment changes estuarine biogeochemistry and given the scale of these modifications, reduction in intertidal/subtidal estuarine area is now potentially a globally significant phenomena which needs to be considered within biogeochemical budgets and deserve further research. The overall impact is likely to be less efficient retention of carbon, nutrients and many other elements and compounds in estuaries, leading to increased eutrophication pressures on coastal waters (Howarth et al., 2011). Furthermore since the processes involved in retention of nitrogen and phosphorus are different, the impacts on the fluvial input ratio of these nutrients are likely to be modified, resulting in changing N/P ratios in coastal waters. Silicon (the other key phytoplankton macro-nutrient) is not retained in estuaries but may be released by dissolution from fluvial sedimentary particles, and therefore the ratio of the nutrients N, P and Si reaching coastal waters will be altered and this can in turn modify the phytoplankton species composition of coastal waters (e.g. Statham, 2012, Howarth et al., 2011). Climate change in coming decades will alter flow rates and seasonality, temperature, wind fields and coastal water current patterns all of which will further modify coastal water biogeochemistry, compounding the stresses of increasing nutrient load and loss of estuarine nutrient retention. Managed realignment of coastal systems to recreate lost intertidal habitat has the potential to restore nutrient retention while also delivering other ecosystem services, particularly in the face of anticipated sea level rise (Luisetti et al., 2014, Turner and Schaafsma, 2015).

**Acknowledgements**

We thank the many students and colleagues who have collaborated in our work on coastal systems. This work is supported in part by the Shelf Sea Biogeochemistry Programme supported by NERC and DEFRA.

We thank two reviewers for helpful comments that greatly improved this manuscript.

We acknowledge the friendship, support, scholarship and joie de vivre of Tom Church which has illuminated our work and play over 35 years.

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