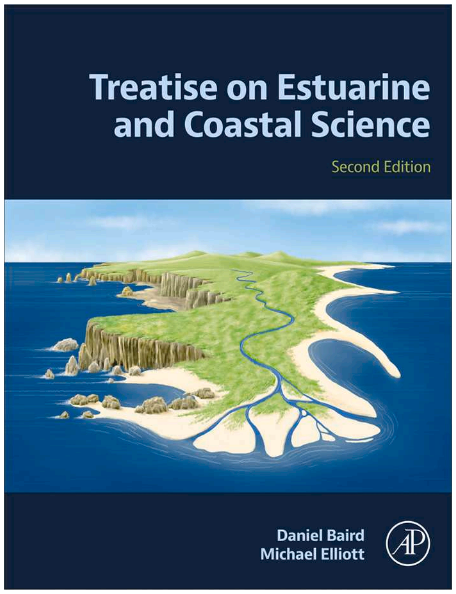

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7.5 Nutrient Recycling and Waste Remediation as a Service From Estuarine and Coastal Ecosystems

Stephen CL Watson and Nicola J Beaumont, Plymouth Marine Laboratory, Plymouth, United Kingdom

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Abstract

The purpose of this chapter is to review environmental and ecological economics research on nutrient recycling and waste remediation of coastal ecosystems. For this purpose, two main questions are addressed: What are the monetary values associated with the ecosystem service of nutrient and waste remediation (NWR)? And, what characterizes good management of this service? It is found that values have mainly been assigned to nutrient treatment by coastal wetlands and shellfish although studies in other types of habitats e.g., kelp, seagrass and sediments are emerging. Removal of nutrients, revealed by a literature comparison of wetlands, ranged between €0–172,227 using a replacement cost valuation approach. It is also observed that nutrient neutrality policies have been implemented in practice for mitigating degradation of coastal ecosystems, but these differ in different parts of the world. The chapter also points at promising future mitigation options using marine nature based solutions, such as bivalve, kelp and seaweed farming.

Key Points

- The ecosystem service of nutrient and waste remediation (NWR) enables humans to utilize the natural functioning of ecosystems to process and detoxify a large number of waste products and therefore avoid harmful effects on human wellbeing and the environment.
- This chapter cites estimates of the key economic values arising from the nutrient removal services provided by a variety of estuarine and coastal ecosystems, including wetlands, seagrass beds and bivalve reefs.
- Discusses direct and indirect valuation studies of coastal zone NWR.
- Outlines management and policy actions to control nutrient inputs from watersheds including ongoing challenges in the UK and European Union.
- Highlights how restoring or creating new structured marine habitats such as saltmarsh, seagrass. kelp and oyster reefs can provide nature-based solutions (NbS) for nutrient management through assimilation, burial and sediment denitrification.

7.5.1 Introduction

Ever since the early studies of salt marsh function (Teal, 1962; Odum and de la Cruz, 1967; Teal and Teal, 1969), coastal areas have been appreciated by ecologists for their functions as highly productive ecosystems. The classic study by Gosselink *et al.* (1972, 1974) attempted to put dollar values on salt marsh functions, including that of waste assimilation (at that time, US$2500 per acre ~ $6175 per hectare). Since then, many other studies have pointed out the key role coastal ecosystems have in recycling nutrient and waste substances. Intact coastal ecosystems, in addition to ranking among the most productive on earth, develop multiple food web linkages that process incoming (allochthonous) materials, thereby transforming organic detritus and dissolved nutrients into new biomass. Due to their abundance within healthy coastal zones, wetlands (e.g., mangroves, saltwater marshes, and seagrasses) and their associated bacterial communities can play a key role in nutrient removal (e.g., Wu *et al.*, 2008; Velinsky *et al.*, 2017), but other taxa, such as filter feeding bivalves (e.g., Broszeit *et al.*, 2016) and other bioturbators and bioirrigators, such as burrowing shrimps or polychaetes, may also take up or remove nutrients via burial in sediments. There has also been growing interest in the potential contributions of shellfish, wetland and kelp cultivation or aquaculture to nitrogen management in the United States, Europe and China (Rose *et al.*, 2014; Dvarskas *et al.*, 2020; Xu *et al.*, 2023).

As described elsewhere in this volume, coastal ecosystems generate several different ecosystem services. Among these, nutrient and waste remediation (NWR) is a key regulating service that is rarely quantified and often goes unnoticed. NWR takes place through ecosystem processes that act to reduce concentrations of wastes by the mechanisms of biological cycling/detoxification and sequestration/storage (Watson *et al.*, 2016). Biological or abiotic transport processes may also act to transport wastes from a given bounded system to another via atmospheric, benthic or lateral export processes (Watson *et al.*, 2016). This service constitutes an input for the production of many provisioning and supporting ecosystem services, such as food (fish, shellfish, and various seaweeds), biodiversity, and recreational values from leisure and bathing. For example, the provision of food such as shellfish will depend upon shellfish stocks, but also upon the service of NWR to ensure clean waters in which the stock can grow. In turn however, the harvesting of the shellfish may have negative implications on the service of NWR as the presence of shellfish plays a fundamental role in the provision of this service.

Threats to the production of this service include excessive eutrophication (hyper-eutrophication) which increases suspended material, reduces light penetration, and limits or slows down biotic uptake. Filter-feeding organisms and wetland habitats will remediate against these effects (thus performing the services of NWR), but in cases of more extensive eutrophication — in some instances causing serious hypoxia (Diaz and Rosenberg, 2011) — the service capacity of the system will be overburdened. Disturbance syndromes may take many years, even centuries, to develop (e.g., Le Moal *et al.*, 2019), but once established, may be difficult to remediate. Filter-feeders and wetlands also assimilate or trap many waterborne wastes, including metals, organic chemicals, plastics and pathogens (Watson *et al.*, 2016). Although metals and persistent organic chemicals can build up to high enough concentrations to have detrimental effects on ecosystem processes (e.g., the impairment of denitrification by metals (Sakadevan *et al.*, 1999), moderate waste loadings of these substances can generally be tolerated by biota without loss of other services.

In this chapter we only consider the nutrients and organic matter removal aspect of NWR (specifically excess nitrogen and phosphorus), however it is important to recognize the ecosystem service of NWR enables humans to utilize the natural functioning of ecosystems to process and detoxify a large number of waste products (including heavy metals, persistent organic pollutants, plastics and pathogens see Watson *et al.*, 2016 for a review) and therefore avoid harmful effects on human wellbeing and the environment. The main objectives of this chapter are to present society's current appreciation of NWR of coastal zones and to investigate actions taken to combat the threats to NWR. This is accomplished by analyzing and surveying studies estimating the value of NWR and presenting potential and actual policies and management actions for combating current threats to the functioning of NWR. As will be evident from the brief survey of valuation studies, there is a lack of such studies applied to NWR by coastal zones, indeed even as compared to the First Edition of the Treatise published in 2011 there has been relatively little progress in terms of understanding and application. This chapter therefore presents novel results from the calculations of such values from the Baltic Sea, which is located in Northern Europe and the coastal zones of the Solent Marine Sites (SEMS) in the UK.

7.5.2 A Framework for Assessing Values of NWR

The ecosystem service of NWR is highly complex to quantify, not least because it is often non-linear and displays thresholds, but also as NWR in one coastal area may affect other waters through marine transport processes. There is no consensus for the assessment and valuation of NWR (Barbier *et al.*, 2011) and a range of options have been applied to calculate their value, including replacement costs (e.g., Dvarskas *et al.*, 2020), choice experiments (e.g. Duijndam *et al.*, 2020), contingent valuation (e.g., Matias Figueroa *et al.*, 2021), opportunity costs (e.g., Matias Figueroa *et al.*, 2021), production functions (e.g., Qian and Linfei, 2012) and damage costs (e.g., Compton *et al.*, 2011). A search of the Ecosystem Services Valuation Database ESVD database (Brander *et al.*, 2023) included a range of NWR valuation methods applied from 1970 to 2019 including: 39 values determined by choice experiment, 19 values determined by contingent valuation, 3 values determined by damage costs, 1 value determined by group or participatory valuation, 5 values determined by market prices, 1 value determined by opportunity cost, 6 values determined by production function, and 46 values determined by replacement costs.

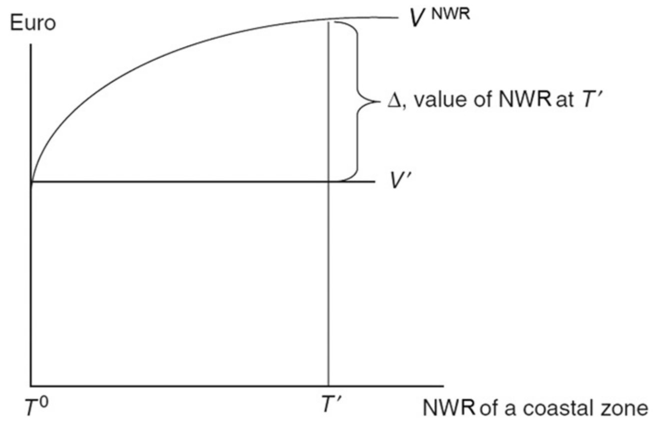

Fig. 1 Illustration of NWR value of a specific coastal zone.

There has also been recent interest in valuing this service from the perspective of developing nitrogen credits (NE, 2023). Increasingly nutrient targets, such as instituted total maximum daily load (TMDL) for nitrogen and phosphorus, are being enforced by regional, national and international authorities and agreements to combat their degrading effects (Compton *et al.*, 2011) on the environment (e.g., Helcom 2007a, Environmental Protection Agency, 2011). Meeting these targets has proved challenging, in part due to the associated costs. A recent response to this challenge has been the development of nutrient trading programs, now well established in the US and increasingly apparent across Europe (Branosky *et al.*, 2011). Originally focused on land-based mitigation practices, such as changing farming practice, credits are now also awarded to environmental sequestration and denitrification, for example by a range of coastal habitats. These programs establish a value per credit, with a credit often equating to a kg of nitrogen, and these values can thus be used as a proxy for the value of the NWR service (DePiper *et al.*, 2017).

Another method of valuing NWR is to value the other ecosystem services which are dependent upon the NWR. In addition to providing a direct benefit increased levels of NWR can lead to higher levels of other ecosystem services delivery as the waters are able to support greater levels of biodiversity, opportunities for wild food provision and aquaculture, and improved recreational and aesthetic benefits (Jones *et al.*, 2014). Burkholder and Shumway (2011) and Irving and Connell (2002) also propose that NWR can increase light penetration due to clearer water allowing marine benthic flora to sequester carbon up to a greater depth than in turbid waters. Building on this concept, the value of NWR can be represented through the valuation of a change in the supply of these associated ecosystem services, as illustrated in **Fig. 1**. The horizontal axis in **Fig. 1** shows increasing levels of treatment, and the vertical axis is monetary measurement. The straight line V' reflects the sum of values of all ecosystem services, including for example food provision, recreational opportunities and biodiversity, without consideration of NWR in the specific coastal zone. The curve V^{NWR} reflects the values obtained when the coastal zone has different levels of treatment capacities by, for example, introduction of increasing number of mussel farms. As illustrated in **Fig. 1**, it is assumed that these values increase at higher treatment levels but at a decreasing rate such that the value approaches an asymptote. The largest value, Δ, is obtained approximately at T′.

Although simple in principle, the assessment of values of NWR in the way depicted by **Fig. 1** is far from a trivial issue in practice. The quantification of these values is obtained in two steps: (1) identification and quantification of all ecological effects of NWR and (2) assessment of these effects in monetary terms. Information is required on the impact of the nutrient treatment on the ecosystem in situ, so-called direct impacts, such as income generated from mussel harvests or improved biodiversity, but also on surrounding ecosystems and dispersal of effects in the entire economy, denoted as indirect impacts. Indirect effects can also occur through the spread of impacts on surrounding water ecosystems, for example, coral reefs are sensitive to nutrient and contaminant pollution (e.g., Done, 1998) and their decline may have a substantial impact on tourism (Finkl and Charlier, 2003).

As discussed in several chapters of this treatise, the challenges to valuation of ecosystem services provided by coastal ecosystems are immensely high. Thus, the valuation approach suggested in **Fig. 1** may not be feasible in practice. Focusing on the replacement cost approach as an alternative, as it is the most prevalent method, this approach uses the cost of replacing an ecosystem or its services as an estimate of the value of the ecosystem or its services. This approach can make use of politically determined targets in a cost-effectiveness framework (see Gren (1999) for application to valuation of land as pollutant sink). This approach is illustrated in **Fig. 2**, where it is assumed that a cleaning target of P^T for phosphorus reductions to a marine water body is determined through setting policy. The curve C shows the minimum costs for achieving different pollutant cleaning targets when NWR is not included as an abatement option in the cleaning program, and C^{NWR} illustrates the minimum costs when it is included. Each point on C and C^{NWR}, respectively, reflects the allocation of all abatement measures that reaches the reduction targets along the horizontal axis at minimum costs. Note that all cleaning levels up to P' are obtained free of charge due to the treatment ability of NWR that takes place without any particular management costs. However, if the cleaning implies deterioration in the coastal zones, this cost should be added to C^{NWR}. For illustrative purposes, let us assume that the P^T illustrates a predetermined target that has been achieved by political decision making.

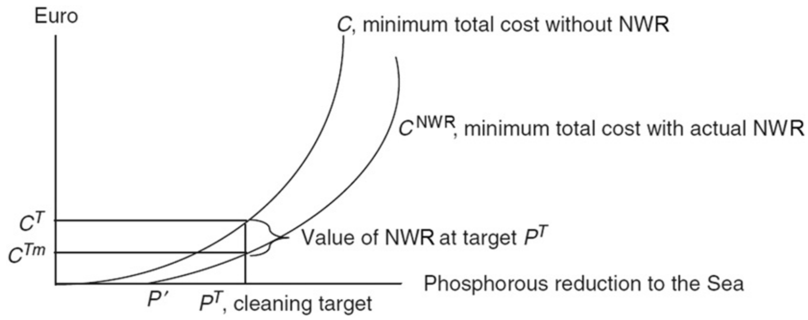

Fig. 2 Illustration of calculation of the value of NWR in a coastal zone as an abatement measure in a cost-effectiveness framework.

The value of NWR at the target phosphorus load target P^T is now determined by the difference in total minimum costs with and without NWR, which corresponds to the distance C^T – C^{NWR} in **Fig. 2**. The value increases from the potential cost advantages of NWR with respect to its multifunctional cleaning capacity, that is, simultaneous cleaning of nitrogen and phosphorus, and the direct impact on the sea. Targets exceeding P^T give a larger value of NWR and targets below P^T generate lower values. However, the value declines if the coastal ecosystem is deteriorated due to the cleaning function.

Although the majority of the monetary values associated with NWR are determined using the replacement cost approach, for example this is the UK Office for National Statistics (ONS, 2021) approach, it is widely considered to problematic due to a number of reasons, including: (1). the replacement cost assumes that the ecosystem service removes all wastes effectively (i.e., there is no residual damage); (2). we need to know how much of the total nutrient and waste loads entering a specific coastal zone are effectively processed by the zone; (3). there is a need for quantification of the ecological impacts on the zone in question from the entering load; (4). the cleaning capacities of the coastal zone need to be commensurate with that of other abatement measures (see Gren, 1999); (5). the costs are unlikely to satisfy the SEEA (System of Environmental Economic Accounting) criteria of being both least-cost and feasible/realistic in the event of ecosystem services loss. Despite this array of difficulties, the replacement cost remains the most frequently applied method as it is relatively simple to deploy and there is an absence of preferable alternatives.

In summary, there are a broad range of methods available for the valuation of NWR, but at present there is no ideal or agreed approach to the assessment and valuation of NWR, and this is a key area for future development. To enable this development, it is recommended to consider in full the methods which have been applied in the past, as discussed in the following sections.

7.5.3 A Brief Survey of Direct and Indirect Valuation Studies of Coastal Zone NWR

Direct and indirect valuation of NWR of coastal habitats has been carried out in an increasing number of studies. This section does not aim to provide a comprehensive list of these studies, but instead, provides an in-depth overview of several habitat and species-specific valuation studies, selected to provide insight to a range of valuation methods, approaches and applications.

7.5.3.1 Direct Valuation Studies of NWR by Wetlands

Wetlands are defined as areas of land that are either permanently or seasonally inundated with water, and include salt and freshwater marshes, swamps and bogs. Although the role of wetlands as pollutant sinks has been acknowledged and extensively researched since the mid-1970s, there are relatively few studies estimating the associated value, although the number of studies per year is steadily increasing over time (Browne *et al.*, 2018). There are, however, many studies estimating general values from improved water quality, such as recreational values, and biodiversity (see Brander *et al.* (2006) for a meta-analysis). In this chapter, we give a brief survey of a few studies aimed at estimating NWR values of wetlands (Gosselink *et al.*, 1972; Breaux *et al.*, 1995; Gren *et al.*, 1995; Byström, 2000; Gren *et al.*, 2009, Watson *et al.*, 2020a). These studies present variation in application regions and scales, and also in estimated results, which are presented in **Table 1**.

Gosselink *et al.* (1972, 1974) used estimates of biochemical oxygen demand (BOD) and nutrient loading into estuarine tidal marshes and calculated the replacement costs of using secondary and tertiary treatments to remove the BOD and nutrients, respectively. Given the early date of that study, such replacement costs were higher than today. Breaux *et al.* (1995) and Gren *et al.* (1995) also provide relatively early examples of studies estimating the values of NWR by wetlands. Breaux *et al.* (1995) estimate replacement value of coastal wetlands for wastewater treatment in three different sites in Louisiana. Wetlands were used as waste treatment facility by three different industries: municipality treatment, food-manufacturing plant, and a seafood processing plant. Comparison of the costs of wetland construction with waste treatment at the factories resulted in NWR values of wetlands ranging between $785 and 9635 per acre and annum. Nitrogen sink values have been estimated for the Danube floodplains, which is shared by eight countries and covering an area of approximately 1.7 million ha (Gren *et al.*, 1995). Simplified estimations were made based on transfers of values obtained by the replacement cost method applied to Swedish wetlands and data on nitrogen

Table 1 Examples of studies valuing nutrient sequestration by wetlands, in 2020 prices (CPI inflation calculator has been used for assessing values in 2020 Euro)

Study and approach	*Application region*	*Results in Euro/ha*
Gosselink *et al.* (1974) replacement cost approach	North America	19,265–172,227
Breaux *et al.* (1995), engineering method	Louisiana as waste treatment for different plants	1427–17,989
Gren *et al.* (1995), transfer of replacement values	Danube floodplains as nitrogen sinks	Germany 273, Austria 296, Slovakia 441, Hungary 737, Croatia 441, Bulgaria 551, Romania 911, Ukraine 1027
Byström (2000), replacement cost approach with estimated nitrogen abatement functions of wetlands	Swedish coastal wetlands as nitrogen sinks	0–621
Gren (2012), replacement cost approach with stochastic programming	Resilience values of Baltic Sea coastal wetlands as nitrogen and phosphorus sinks	Denmark 4062, Finland 12, Germany 215, Poland 52, Sweden 52, Estonia 122, Latvia 1, Lithuania 12, Russia 12
Watson *et al.* (2020a), replacement cost approach	UK's Solent Marine Site (SEMS) region	126,722 for nitrogen 15,761 for phosphorus

sink capacities by the Danube floodplains. The differences in results among countries are then mainly determined by wetland cleaning capacities as well as by gross domestic product (GDP) growth during the years 1995–2008.

The Byström (2000) study is the most profound with respect to estimation of the production function for NWR by wetlands. Econometric estimates were made based on data on nitrogen cleaning from a number of Swedish wetland construction programs carried out during the 1990s. The value of wetland as a cleaning option was then calculated as the savings in costs from replacement of more costly measures within the agricultural sector at different nitrogen reduction targets for Sweden. Positive replacement values occurred at levels exceeding 30% reductions in nitrogen loads to the Swedish coastal water. At maximum, which occurred at a 50% reduction of nitrogen load, the estimated replacement value of wetlands amounted to 5390 ha^{-1}.

Gren (2019); Gren and Limburg (2012) addressed the role of wetland when damages from pollution are uncertain and calculated the so-called resilience values of coastal wetlands in the Baltic Sea drainage basin. Given environmental targets set by policymakers, resilience value was then estimated as the value of changes in the reliability of reaching the predetermined target(s) for maximum loads of nitrogen and phosphorus. Reliability was, in turn, assessed by means of risk measurement where improvements in reliability, or in resilience, were determined by changes in total risk from different actions available to the decision makers. It was shown that resilience values are positive when the nutrient load from emission sources and wetland nutrient abatement capacity are positively correlated. Total risk in nutrient load then declines due to wetland construction, which in turn, implies that the predetermined nutrient targets are achieved with a higher probability. The results presented in **Table 1** are evaluated at 20% decreases in nutrient and a reliability level of 0.975, that is, minimum probability for achieving the targeted nutrient reductions was set at 0.975. Difference results among countries were explained by the variability in loads from emission of sources and costs of other abatement measures than wetland construction. As shown in **Table 1**, the value is particularly large for Denmark where nutrient loads, mainly from the agricultural sector, are highly stochastic and abatement costs for this sector are relatively large.

Watson *et al.* (2020a,b)mapped saltmarsh, along with six other key temperate coastal habitats (littoral sediment, mat-forming green macroalgae, subtidal sediment, seagrass, reedbeds see Section "7.5.4.5"), within the UK's SEMS. They then estimated the capacity of these habitats to remove nitrogen and phosphorus drawing on previous studies by Adams *et al.*, 2012 and Blackwell *et al.*, 2010. This research reported that methods to determine rates of nitrogen and phosphorus bioremediation in saltmarsh were highly variable, with varying aspects assessed and making standardized extrapolation challenging. They also note that rates are influenced by a number of factors including season, local hydrology regimes, nutrient loading rates and the balance of population/ habitat level processes (e.g., photosynthesis, respiration and dissolution). As such, they report an overview of the published rates but with this potential uncertainty acknowledged. For the saltmarsh habitats in the Solent the biophysical change of nitrogen was documented as 475 tonnes/ha/yr and for phosphorous 62 tonnes/ha/yr, this when coupled with replacement costs the value/ha/ yr, was documented as £111,009/ha/yr for nitrogen and £13,807£/ha/yr. When combined with the area of saltmarsh in the SEMS, this resulted in values of £ 139.9 million per year for nitrogen and £17.39 million for phosphorus. For nitrogen, the saltmarsh was found to be the coastal habitat with the highest value for this service and for phosphorus it was the second highest following littoral sediments with macroalgae.

7.5.3.2 Direct Valuation Studies of NWR by Shellfish

There has been growing interest in the potential contributions of shellfish aquaculture and natural bivalve reefs to nutrient management. Suspension feeding, and specifically filter feeding, by shellfish plays a key role in NWR. By actively pumping large volumes of water over a filter, shellfish remove plankton and detritus and thus improve water quality. As this filtered material is used for nutrition the nutrients are then incorporated within the tissues and shell, and when the shellfish are harvested the nutrients are thus removed from the local environment. Filter feeders, such as mussels, clams and oysters, are often found in dense populations and as such their activities can have a substantial impact on water quality, reducing effects of eutrophication as well as sediment, harmful bacteria and contaminants (Dame *et al.*, 1984; Hoellein and Zarnoch, 2014).

Similarly to wetlands, the majority of the valuation studies apply a replacement cost approach (Beseres Pollack *et al.*, 2013; Bricker *et al.*, 2020; Watson *et al.*, 2020a). Beseres Pollack *et al.* (2013) focused on the existing oyster population and used field-based estimates of nitrogen removal rates in the Mission-Aransas Estuary coupled with a cost for the equivalent wastewater treatment plant (WWTP) biological nitrogen removal, resulting in an estimated value for the nitrogen removal services of oysters in the Estuary of US$113 471/year. Bricker *et al.* (2020) also applied the replacement cost approach to value NWR by oyster aquaculture in Long Island Sound. Costs associated with WWTP and agricultural or urban best management practices (BMPs) were applied resulting in values of between US$8.5 and 230 million per year, with the variation driven by the replacement technology selected and acreage covered. Following a similar method, and as noted in the previous section, Watson *et al.* (2020a) used a range of regionally relevant replacement cost values of 295 [£/kg] for nitrogen and 282 [£/kg] for phosphorus. For the native oyster (Ostrea edulis) habitats in the SEMS, the biophysical change of nitrogen was documented as 123 tonnes/ha/yr and for phosphorous 25 tonnes/ha/yr, thus when coupled with the replacement costs the value/ha/yr was documented as £12,774/ha/yr for nitrogen and £2483£/ha/yr for phosphorus. When combined with the area of oyster beds in the Solent, this resulted in values of £37.44 million per year for nitrogen and £6.77 million for phosphorus.

Other methods in addition to the replacement costs have also been applied. For example, a nutrient credit-based valuation has also been undertaken for Oysters in North Carolina (Grabowski *et al.*, 2012). This approach multiplies the value of a nitrogen nutrient credit from an existing, relevant trading program by the expected nitrogen removal or sequestration. Applying this method resulted in an estimation of benefits of nitrogen removal by an oyster reef to be in the range of $1385–6716/ ha/yr. DePiper *et al.* (2017) used a similar methodology to value a restored oyster reef on the eastern shore of Maryland. The restoration project cost is estimated to be over US$31.65 million. Permitting harvesting, but not including nutrient credits, the NPV of the oyster reef is US$8.67 million, however if nutrient trading is allowed (i.e., the value of the NWR is included) the NPV increases by US$1.99 million. Including the NWR value leads to a situation where the restoration project is estimated to break even after an average of 18 years, depending on the specifics of the scenario. Given the high initial costs and delayed profits it is unlikely that private funding will be viable, but the inclusion of the NWR value does make the restoration more likely to be government-funded oyster reef through bonds repaid with income generated by the reef. The authors also note that the oysters will provide a range of other benefits which if also valued would likely add to the financial viability of the restoration project.

Interestingly, Dvarskas *et al.* (2020) applied both a replacement cost approach and a nutrient trading approach to value Clam and Oyster Annual Nitrogen Removal in Greenwich Bay, Connecticut. They include a more sophisticated modification to the replacement cost approach, namely an allocated solution approach which assigns potential NWR replacement costs for clam and oyster in a way that is constrained to the real-world options available to watershed resource managers. The results found that the basic replacement cost method of using WWTP upgrades gave an annual value of US$100,871, the allocated solution approach resulted in a higher value of between US$2,315,829 and US$5,795,449, and the nitrogen credit valuation approaches indicate US $206,448. This research clearly demonstrates how the valuation method applied can make a significant difference to the results, and in turn if these figures are applied in a policy setting this can also have substantial influence in decisions which are made around coastal habitat restoration and conservation. Finally, (Gren, 2019) researched the potential for mussel farming as a low-cost and equitable option for mitigating damage from eutrophication in the Baltic Sea, applying the costs of nutrient removal by mussel farming as derived by Gren *et al.* (2009) and which as above applied a replacement cost approach to estimate the value of the NWR service.

7.5.3.3 Direct Valuation Studies of NWR by Other Coastal Habitats and Species

Although the majority of the NWR literature is focussed on wetlands and shellfish, other coastal habitats and species are also documented to deliver this service, including seaweeds such as kelp, seagrass or eelgrass, fisheries and sediments. The research into the NWR values of seaweeds (including kelp) is still nascent, with little published in this area. Kim *et al.* (2014, 2015) estimated nitrogen removal by kelp farm systems using experimental data. They then multiplied these estimates by the value of a nutrient credit in Connecticut (CT), calculating annual values for nitrogen sequestration of between US$147 and US$1226 per hectare. This range in values was driven by variability in the and location of the farm.

Research into the NWR values of sea (or eel) grass is similarly minimal. Cole and Moksnes (2016) identify and quantify links between three eelgrass functions, habitat for fish, carbon, and nitrogen uptake. They find that over a 20-year period a hectare of eelgrass, including the organic material accumulated in the sediment, sequesters 466 kg nitrogen as compared to unvegetated habitats. They apply a regionally derived replacement cost of US$21.3/kg nitrogen to then value the NWR service. This results in an estimate of the value of nitrogen storage per hectare of eelgrass to be approximately US$ 9280 over 20 years, or US$ 680 annualized. Of the three services assessed, the NWR constituted 46% of the total value. A further study on seagrass by Moksnes *et al.* (2021), again using a replacement cost method and building on Cole and Moksnes (2016), found that eel grass loss has resulted in a release of 6.63 Mg nitrogen per hectare, with an estimated economic cost to society of 141,355 US$/ha.

Nielsen *et al.* (2019) researched the economic value of NWR of Danish, Finnish and Swedish pelagic fisheries using a shadow value of the abatement cost in alternative sectors, as derived by Gren *et al.* (2008). This research calculates the cost of removing 1 kg nitrogen is €3 and the cost of removing 1 kg phosphorus is €67, equating to a total value of €0.37 per kg herring and sprat removed from the Baltic Sea. This value was then applied in a dynamic bio-economic model, FishRent, to explore the impact of various policy scenarios. Under these different scenarios the NPV of nutrient reduction was found to vary between 1123 million € and 1515 million €. The outcome of the models shows that maximizing catch volumes while having a flexible system for quota trade within the fishing sector results in the highest social welfare gain.

7.5.3.4 Indirect Valuation Studies of Eutrophication Mitigation

Like the direct valuation studies of NWR, studies of indirect valuation of NWR as reductions in eutrophication vary with respect to scale and methods. A large number of studies have been applied to the Baltic Sea (Söderqvist, 1998; Sandström, 1999; Söderqvist and Scharin, 2000; Soutukorva, 2001), with also a focus on the eutrophied freshwaters in the UK (Pretty *et al.*, 2003) and in the US (Dodds *et al.*, 2009), and harmful algal blooms in the US coastal waters Hoagland *et al.* (2002). The studies applied to the Baltic Sea use different valuation methods, the contingent valuation method (CVM), the travel cost method (TCM). The other three studies apply the method of benefits transfers. It is notable a search of the ESVD database between 1970 and 2019 found that after replacement costs (n = 46) choice experiments (n = 39) and contingent valuation (n = 19) were the second and third most common method of NWR valuation, and important to consider that, whilst equally valid, these methods provide values which are not commensurate with the replacement costs due to the different theoretical foundations of the methods. The studies also use different valuation scenarios, and their results are difficult to compare (see **Table 2** for a listing of study, valuation method, application region, and results).

Five of the studies estimate values at the national scale (Gren *et al.*, 1997; Hoagland *et al.*, 2002; Pretty *et al.*, 2003; Dodds *et al.*, 2009) and the other three are applied to regional scales in Sweden. The Gren *et al.* (1997) study carried out contingent valuation studies in Sweden and Poland of the willingness to pay for returning to the conditions of the Baltic Sea prevailing in the 1950s. The results from the Swedish study are transferred to Finland, Denmark, and Germany (see (Söderqvist, 1996)). Estimated values from the Polish study are transferred to the remaining riparian countries. In Hoagland *et al.* (2002), the impact of nutrient loads on harmful algal blooms is estimated. They include several types of effects: on public health, commercial fisheries, recreation, and tourism. The impacts of harmful algal blooms on the different ecosystem services were quantified by means of expert judgments. The monetary correspondences were assessed by inquiries to affected industrial sectors and by benefit transfers from other studies. A similar approach was applied in Pretty *et al.* (2003), the study of which covered a range of ecosystem services affected by eutrophication: property values, drinking water quality, recreational values, tourism, and ecological effects on biota. Dodds *et al.* (2009) attempt to estimate damage costs of several ecosystem services impacted by eutrophication of US freshwater systems. They include effects of eutrophication on recreational values, property values, and biodiversity. These monetary estimates are obtained from calculations of eutrophication production functions with respect to the included ecosystem services, each of which are assigned values by means of benefit transfers from other studies.

All of the remaining three studies are applied to different coastal regions in Sweden using the CVM and the TCM. A TCM study measures only the so-called user value of the good in question, that is, recreational value of the archipelago, and does not include nonuse values such as option and/or existence values. The CVM includes both use and nonuse values, which explains the difference in results between the studies. However, studies applying CVM are subjected to much criticism concerning their hypothetical way of estimating values (see, e.g., (Turner *et al.*, 2003)). The CVM is used by Söderqvist and Scharin (2000) and the

Table 2 Examples of studies estimating values from mitigation of eutrophication, in millions of 2020 Euro (CPI inflation calculator has been used for assessing values in 2020 Euro)

Study, valuation method	*Scenario and region*	*Results, annual value in millions of Euro*
Gren *et al.* (1997), contingent valuation method	Return to a 'healthy' Baltic Sea prevailing in 1950s	Denmark 894, Finland 570, Germany 501, Poland 924, Sweden 1358, Estonia 44, Latvia 60, Lithuania 59, Russia 471
Sandström (1999), travel cost method	Sight depth. Laholm Bay at the Swedish west coast	3 in average
Söderqvist and Scharin (2000), contingent valuation method	1 m sight depth improvement. Stockholm Archipelago	62–104 per m sight depth increase
Soutukorva (2001), travel cost method	1 m sight depth improvement. Stockholm Archipelago	7–14 per m sight depth increase
Hoagland *et al.* (2002), expert judgments, benefit transfers	Costs of harmful algal blooms in the US coastal waters	55 in average
Pretty *et al.* (2003), benefit transfers	Freshwater eutrophication in the UK	145 in average
Dodds *et al.* (2009), benefit transfers	Damage costs of human induced eutrophication of US freshwater systems	1834 in average

TCM approach is applied by Sandström (1999) and Soutukorva (2001) for estimating individuals' willingness to pay (WTP) for an improved sight depth in Laholm Bay at the Swedish West coast and in the Stockholm archipelago.

7.5.3.5 Comparison of Results

The reported studies of direct and indirect valuation use different scenarios for valuation, methods, and application regions. As such the comparison of the values is very challenging as values derived by different methods will be non-commensurate. Authors also report the values in different ways with some reported as value/ha, others a value/ha/year, and others as value/area. The stated preference studies have a different format again with values in per person or per household format. Transfer of these values between sites and species is also challenging as most values will be site specific. Replacement costs in many studies have been derived to be regionally or nationally specific and so cannot be readily transferred. It is thus advised not to directly compare the results, but rather to consider the values as individually relevant.

7.5.4 Some Case Studies

The purpose of this section is to show in more detail how the replacement cost methods described in Section "7.5.2" have been applied to a range of specific national (Baltic Sea Europe) and regional (Solent Marine Sites UK) ecosystems and coastal zones.

7.5.4.1 Values from Reductions in Nutrient Pollution Baltic Sea

The Baltic Sea is located in Northern Europe and is the world's largest brackish marine waterbody. It provides multiple ecosystem services to people from the nine countries which border it. However, the flow of these services is currently threatened by poor water quality in many areas of the Baltic. Previous work has applied mainly nutrient abatement costs (e.g., Gren *et al.*, 2009) and stated preference approaches (e.g., Czajkowski *et al.*, 2015) to estimating the economic benefits of polices which reduce nutrient inflows to the Baltic at the level of individual nations.

One often-used criterion is that of cost-effectiveness, which is defined as the allocation of abatement measures in different countries which generates the target at the minimum overall cost. The condition for this is that marginal costs of all measures are equal. As long as marginal costs differ among measures, it is always possible to reallocate abatement and obtain the same target at a lower cost. This is made by reducing cleaning at the relatively high-cost measure and increasing it by the same amount by the low cost measures.

Marginal costs of nutrient reductions to the Baltic Sea or any of its basins consist of two main parts: cost for the measure and its impact on the Sea target. Starting with the cost for a specific abatement measure, say improved cleaning at sewage treatment plants or reductions of fertilizers, it is most often less expensive to clean up nutrients at low cleaning levels. At higher cleaning levels, it becomes increasingly more costly to clean up another ton of nitrogen or phosphorus. Such a typical shape of the marginal cost for cleaning at the emission source is illustrated in **Fig. 3**.

The *MC* curve in **Fig. 3** illustrates how cost for cleaning of an additional ton of nitrogen is increasing for higher cleaning levels. Each point on the curve shows the minimum cost for an additional cleaning by 1 ton. It is then assumed that the firm uses its resources for cleaning, such as labor and capital, in order to minimize total cleaning cost at each cleaning level. If this is not the case, such as under the requirement of best available technology, the marginal cost becomes higher for larger cleaning levels. In practice, however, it is difficult to estimate such a smooth marginal cost function as illustrated in **Fig. 5**. Instead, constant marginal abatement cost, or unit costs, are used to compare costs of different measures (see, Hautakangas *et al.*, 2014). This implies a horizontal line in **Fig. 3**. However, the curve illustrated in **Fig. 3** is, for most emission sources, not the marginal cost for reaching nutrient reduction targets in the Baltic Sea. In order to find this marginal cost, we need to specify the target and to calculate the effect on the target for the specific emission source illustrated in **Fig. 3**. Let us assume that the target is specified as maximum loads to the coastal waters of the Baltic Sea. The curve in **Fig. 3** is then the same as the marginal cost for achieving the target for a source located at the coastal waters with direct discharges into the sea, a so-called point source. However, for nonpoint sources located

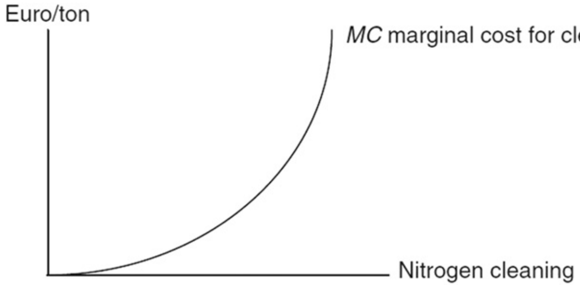

Fig. 3 Illustration of marginal cleaning cost at an emission source.

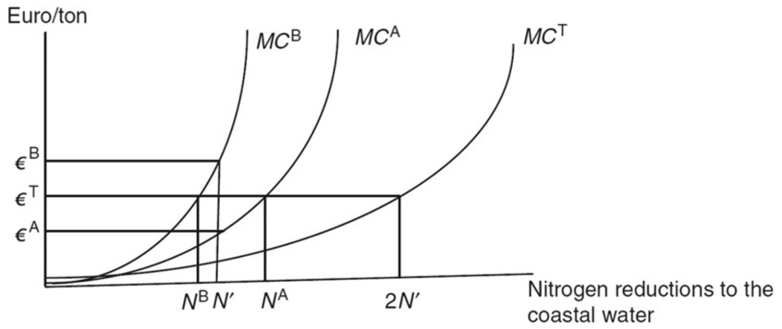

Fig. 4 Illustration of marginal costs for nitrogen reductions to the coastal waters from treatment plant A located at the coast and an upstream treatment plant B (MC^A, MC^B, T M^C marginal cost for plant A, B, and for both plants; 2 N′ nitrogen reduction target; N′ requirement of equal cleaning at the plants; N^A, N^B cost-effective cleaning by plants; €^{A}, €^{B} marginal costs for plants A and B at N′; €^{T} marginal cost at cost-effective cleaning).

upstream in a drainage basin, the marginal cost at the emission source needs to be combined with the effect on the target from the source in order to represent the marginal cost for nitrogen reduction to the Sea. This is illustrated in **Fig. 4** where we have two identical emission sources, that is, a sewage treatment plant located at the coast, A, and an identical plant located upstream, B. Their marginal costs at the emission sources are the same and correspond to the MC curve in **Fig. 3**. However, due to the upstream location of plant B, only half of its effluents reach the coast. This means that it becomes twice as expensive to reduce 1-ton nitrogen to the coast from plant B as compared to plant A.

Let us illustrate this important difference in impacts between the two plants with a simple numerical example, where we assume that the marginal cost is 1 Euro per kg nitrogen reduction for both plants. The nitrogen retention rate for the upstream-located plant B is assumed to be 0.5. The marginal cost for 1 kg N reduction to the sea is now determined by the marginal cost at the source divided by the impact. The impact of 1 kg N reduction from plant A is 1, and for plant B it is 0.5. This means that the marginal cost for N reductions to the sea from plant A is Euro 1/kg N reduction and from plant B Euro 2/kg N reduction. The larger the impact for a given marginal cost at the source, the lower is the marginal cost of N reductions to the sea, and vice versa. Marginal costs for different nitrogen reductions to the coast for the two plants are illustrated in **Fig. 4**.

Note that, due to the specification of the target, the horizontal axis in **Fig. 4** shows nutrient reductions to the coastal waters, and not emission reductions at a source as in **Fig. 3**. At N', the two plants clean the same amount, but their marginal costs such that €^{B} > €^{A}. We can then keep the same total level of cleaning and obtain cost savings by increasing cleaning for plant A and decreasing cleaning for plant B, which, at N'', gives a net saving of €^{B} – €^{A} for a switch by 1 ton of nitrogen. Obviously, the cleaning allocation where each plant abates *nitrogen*′ is not a cost-effective allocation of abatement.

The curve MC^T shows the marginal costs for abatement by both the plants, which is the sum of abatement of each plant for different levels of *MC*. The target under the individual quotas of N' corresponds to $2N'$. It is shown in **Fig. 4** that the marginal cost of €^{T} is the same for both the plants, and it is then not possible to redistribute cleaning among the plants and obtain the total cleaning of $2N'$ at a lower cost. Thus, the cleaning allocation of N^A and N^B gives a cost-effective solution.

The determination of cost-effective allocation of measures as illustrated in **Fig. 4** requires information and data on impact on the predetermined target(s) and cost of the abatement measures in question. Starting in the early 1990s there are a number of studies estimating costs of nutrient reductions to Baltic coastal waters (see Gren and Säll 2015 for a review). Costs of nutrient loads have been estimated in several studies (Gren *et al.*, 1997; Wulff *et al.*, 2014; Gren and Destouni, 2012; Gren and Elofsson, 2017). The Gren *et al.* (1997) study includes one of the earliest studies, which implies that nutrient treatment values of separate nitrogen and phosphorus reductions are in the same order of magnitude, and amount to approximately 900 million Euros at the 50% reduction levels. However, the nutrient treatment values are unequally divided among countries, in absolute values, per capita, and in relation to GDP. This can be seen from **Table 3** where these parameters are calculated for the achievement of the Baltic Sea Action Plan (BSAP) targets (Helcom, 2007b). The plan suggests overall reductions corresponding to 25% of nitrogen and 54% of phosphorus. Poland obtains the largest share, 0.54, of the total value of nutrient cleaning. However, when instead relating the value to population size and GDP, Estonia has the largest values. This is partly due to the large cleaning capacity of coastal zones in the Gulf of Finland, and also due to the savings of costs from decreased needs for relatively expensive measures.

A study by Gren and Elofsson (2017) further develops the Baltic Sea model presented in **Table 3** by evaluating four different schemes for credit trading in nutrient loads to the Baltic Sea. The article examines theoretically and empirically different nutrient trading market designs: (1) with and (2) without credit stacking, (3) a market for a bundled payment of nutrients, and (4) separate markets for either nutrient. The first market, with stacking of nutrient credits, implies that a measure with impacts on both nutrients, such as wetland construction, is allowed to sell credits in both nitrogen and phosphorus markets. Where no stacking of nutrient credits is allowed credits can be sold in either the nitrogen or the phosphorus market, but not both. In the third market type, the bundled nutrient market, nitrogen, and phosphorus abatement are given certain weights and converted into a common nutrient currency (e.g., the Redfield ratio of nutrients (7:1 for N:P by weight) for balanced growth of algae) before being traded on a single nutrient market. The final type of market constitutes a choice of either a nitrogen or a phosphorus market. In this case, the

Table 3 Allocation of nutrient treatment values among Baltic Sea countries for the achievement of the BSAP targets of 25% nitrogen and 54% phosphorus reduction from Gren *et al.* (2013)

Country	*Value in million Euros*	*Value per capita, Euros*	*Value as % of GDP*
Denmark	23	5	0.01
Estonia	83	62	0.74
Finland	121	23	0.08
Germany[a]	9	3	0.01
Latvia	52	23	0.38
Lithuania	63	19	0.32
Poland	602	16	0.26
Russia[a]	99	11	0.24
Sweden	61	7	0.02
Total	1113	15	0.12

[a]Values per capita and as per cent of GDP are based on population and GDP in the drainage basins of the Baltic Sea
Source: See **Table A1** in Appendix

abatement nutrient abatement measures are restricted to the achievement of the target of chosen nutrient, such as in **Table 3**. A key finding of Gren and Elofsson (2017) is that the total abatement cost of achieving reduction targets of both nutrients is always lowest if a market design with credit stacking is established. The application to the Baltic Sea shows that the total abatement cost can be 20% higher (cost difference of 722 million Euro) when credit stacking is not allowed than when it is allowed (Gren and Elofsson, 2017). In comparison, the outcome of bundled market outcomes were more varied and could be higher or lower than markets without credit stacking depending on the value of the ecological improvements created by the excess abatement in relation to the additional abatement costs (see also Gren (2019).

7.5.4.2 Valuation of Coastal Zone Nutrient Treatment in the Solent Marine Site UK

The SEMS covers several harbors, estuaries, areas of open coast and inshore water around the Solent coastal area of UK, many of which are designated as internationally important sites for conservation. These estimates are based on a replacement cost model of nutrient reductions to the SEMS, which is briefly described prior to the presentation of the estimated values of NWR of different coastal zones in the SEMS.

7.5.4.3 Brief Presentation of the Replacement Cost Model

Costs of abatement measures implemented in any of the drainage catchments of the SEMS are determined by their impacts on the target set for the SEMS and on the abatement cost at the location of the measure. Impacts of measures implemented in the catchment depend on nutrient loadings in the catchments, which, in turn, are determined by emissions from sources, leaching, and retention during transport from the source to the coastal waters. Since these transport factors differ among different regions in the catchments of the SEMS because of variation in climatic, hydrological, and biological conditions, the entire Marine Site is broken down into twelve smaller units, known as Water Framework Directive (WFD) water bodies. These are made up of reaches or entire lengths of designated watercourses, with nutrient loads into each of the water bodies. Nutrient transports from sources and costs of abatement measures are calculated for each of these water bodies, which are briefly presented in this chapter. Unless otherwise stated, all data and calculations are found in Watson *et al.* (2020a,b).

In the SEMS, pollution from nutrients, mainly comes from agriculture (50%) and coastal background sources (40%) with only 10% estimated to originate from urban discharges. Nitrogen and phosphorus loads to the SEMS are therefore divided here into three main classes: urban emissions, agricultural loads, and discharges of sewage from households and industry, which is sufficient for sources with direct discharges into the SEMS, such as industry and sewage treatment plants located along the coast. For all other sources (e.g., airborne emissions) further information is needed in the SEMS on the transformation of nutrients from the emission source to the coastal waters. This requires data on transports of airborne emissions among drainage basins, leaching and retention for all sources with deposition on land within the drainage basins, and on nutrient retention for upstream sources with discharges into waterways.

The impact of in treated sewage effluent and nutrient loads from arable land on the receiving watercourses and coastal waters was modeled for each water body using the Environment Agency's Simcat or River Quality Planning (RQP) toolkits for the 2019 period. Estimation of discharges of nutrients from households is based on annual emission per capita in different regions, and on connections of populations to sewage treatment plants with different cleaning capacities. Deposition of nutrients from arable land then includes manure and fertilizers. It is assumed that remaining nutrients from households and industry in the water bodies are discharged into rivers and streams, and the final deposition into the SEMS then depends on nutrient retention. Given all assumptions, the calculated total nutrient loads of approximately 5016 tonnes of nitrogen (nitrogen in the form of dissolved inorganic nitrogen) and 602 tonnes of phosphorus (phosphorus in the form of dissolved inorganic phosphorus) enters the SEMS.

Table 4 Nutrient loads and coastal retention for SEMS waterbodies

Water body	*Total nitrogen loading (kg yr^{-1})*	*Total phosphorus Loading (kg yr^{-1})*	*Relative loading proportion nitrogen (%)*	*Relative loading proportion phosphorus (%)*	*Nitrogen coastal retention (%)*	*Phosphorus coastal retention (%)*
Lymington Estuary	145,030	2370	2.89	0.39	63	100
Beaulieu Estuary	121,280	1140	2.42	0.19	75	100
Southampton Water	1520,106	244,870	30.30	40.65	22	19
Hamble Estuary	198,128	19,270	3.95	3.20	27	100
Portsmouth Harbor	800,249	76,028	15.95	12.62	40	100
Langstone Harbor	370,749	31,276	7.39	5.19	99	100
Chichester Harbor	1275,378	185,089	25.42	30.73	43	88
Pagham Harbor	225,702	23,171	4.50	3.85	37	47
Yar Estuary	37,950	2279	0.76	0.38	70	100
Newton Harbor	133,247	9597	2.66	1.59	49	100
Medina Estuary	117,635	3656	2.34	0.61	20	100
Bembridge Harbor	71,017	3588	1.42	0.60	16	100
Total	5016,471	602,334				

Data on nutrient retention by coastal zone habitats were obtained by Watson *et al.* (2020b) who reports retention for the twelve different water bodies. Data on loads and retentions for the different water bodies are presented in **Table 4**.

Southampton Water is the largest contributor of both nitrogen and phosphorus loads to the coastal waters of the SEMS, accounting for approximately 30% and 40% of total nitrogen and phosphorus loads, respectively. The Hamble Estuary also discharges nutrients only into Southampton Water, which implies that these waterbodies have only one number on coastal retentions of nitrogen and phosphorus. All other water bodies (with the exception of Pagham Harbor) discharge into the open water Solent Strait, and their retentions, as measured in shares of nutrient load entering the respective coastal zone, differ.

The last two columns in **Table 4** show the potential for coastal habitats to retain nutrient loads entering the respective water body. In total, this implies a reduction of nitrogen and phosphorus loads into the recipient waters by approximately 40% and 85% respectively. Thus, a considerable share of total nutrient loads to the SEMS is cleaned up by the coastal zones free of charge. Regional estimates for the proportion of the nitrogen loads that could potentially be removed by habitats varied widely between the water bodies, ranging from 16% in Bembridge Harbor to 99% in the case of Langstone Harbor. Estimates of phosphorus removal from the land-margin showed that net burial in sediments by habitats appears to remove more phosphorus from every region of the SEMS (i. e., 100%) except; Southampton Water (19% removed), Chichester Harbor (88% removed) and Pagham Harbor (47% removed).

The replacement cost model includes 17 different measures for nitrogen reduction and 8 abatement measures for phosphorus based on actual costs of nutrient reduction measures undertaken on the UK's southeast coast. Since diffuse nutrient loadings from the agricultural sector and coastal background sources accounts for approximately 90% of nitrogen and phosphorous loads, the majority of the abatement measures were chosen based on a combination of UK nutrient management and planning documents (Bryan *et al.*, 2013; RSPB, 2013; BPPDC, 2017) (see **Table 5** for a list of included abatement measures). Included measures are catchment sensitive farming approaches (CSF), payments for ecosystem services schemes (PES) and costs involved with upgrades to existing wastewater treatment plants. For each abatement measure, costs are calculated which do not include any side benefits, such as provision of biodiversity by wetlands or additional carbon sequestration and storage. Furthermore, abatement measures located in the water bodies may have a positive impact on water quality, not only in the SEMS, but also in ground and surface waters. However, such data on side benefits are not available for the included abatement measure. This implies an over-estimation of net abatement costs of measures implemented in the water bodies. On the other hand, the cost estimates do not account for dispersion of impacts on the rest of the economy from implementation of the measure in a sector, such as possible increase in prices of inputs of a simultaneous implementation of improved cleaning at sewage treatment plants. Market prices are also used for assessing the costs of conversion of arable land into land uses with lower loading potentials such as wetlands and buffer strips. However, there are insufficient data to evaluate the effect of massive land conversion on the market price of arable land, and constant prices of land are assumed for the cost calculations. For a detailed presentation of abatement capacities and costs of all measures, the reader is referred to Watson *et al.* (2020b).

Table 5 Abatement measures included in the replacement cost model

Measure	*Nitrogen reduction*	*Phosphorus reduction*
Catchment sensitive farming (CFC)	Application of CSF across whole catchment	Regulatory controls on agricultural phosphorus
	Establishment of cover crops following winter wheat	Storing and transporting excess phosphorus from dairy farms to arable farms
	Baling and removal of Oilseed Rape straw	Make available compost to improve soil condition
	Moving from Oilseed Rape to spring beans	
	Move from Oilseed Rape to winter oats	
	Use of clover in place of nitrogen fertiliser on all managed land	
	No tillage and reduction in livestock numbers to achieve 100% nitrogen reduction	
	10% reduction in fertiliser applied to oilseed rape	
	Reduced 20% application of nitrogen to managed grassland	
	Allow field drainage systems to deteriorate including land adjacent to watercourses, natural wetlands and ribbon areas.	
Payments for ecosystem services (PES)	Local conservation body purchases farm holding and over time changes land use	Change in land use from intensive to less intensive grass production
	Provide grants for farmers to change land use to commercial woodland	Creation of wetlands
	Change of use of public owned land from agriculture to sparsely treed landscape.	Taking out agricultural land (arable or grass) Production through offsetting
	Purchase and reversion (ceasing fertiliser use) of arable land	
	Purchase and reversion (ceasing fertiliser use) of managed land	
	Land and change use to sparsely treed sparsely treed	
Upgrades to existing wastewater treatment plants and associated drainage infrastructure	Improve the discharge quality at treatment plants *via* Introduction of nitrogen stripping measures.	Reducing flows through sewage network through water efficiency measures
		On site treatment with disposal systems (e.g., phosphorus stripping or wetlands)

7.5.4.4 Estimated Values of NWR by Different Coastal Zones

As reported in Section "7.5.4.1", the value of nutrient treatment by coastal zones depends on the cleaning capacity, costs of alternative abatement measures, and the chosen nutrient load targets. Two UK ministerial agreements on nutrient load targets have been proposed, one to reduce nitrogen, phosphorus and sediment pollution from agriculture into the water environment by at least 40% by 2038 (HM Government, 2023). The latter agreement aims to reduce phosphorus loadings from treated wastewater by 80% by 2038. However, since the role of nutrient loads for damages from eutrophication in the SEMS is still unclear (Watson *et al.*, 2022), we present results from calculations of values for different targets of nitrogen and phosphorus, respectively, and also for simultaneous reductions in both the nutrients. In addition, we present the implications of these treatments for different water bodies. Recall from Section "7.5.4.1" that the nutrient treatment value of the coastal zone is calculated as the difference in minimum costs for a given nutrient target with and without NWR. Estimates of such values are presented in **Fig. 5**, for reductions targets up to the 100% levels from the reference loads in **Table 4** are presented.

As shown in **Fig. 5**, the nutrient treatment values of separate nitrogen and phosphorus reductions are in different orders of magnitude, and amount to approximately £960 million and £180 million at the 50% reduction levels. The corresponding value of simultaneous reductions, approximately £9200 million, at 100% loadings is much higher due to the high wastewater infrastructure costs associated with treating nutrients and wastes to a tertiary level of cleaning. The lower cost, up to 50% of loadings, is due to the multifunctional characteristic of several abatement measures, in particular land-use changes, which abate both nitrogen and phosphorus at the same cost of land use.

However, the nutrient treatment values are unequally divided among the water bodies. These valuations can also be disaggregated to provide nutrient reduction values (£) specific to different regions in the SEMS. **Table 6** provides a summary of the value of nitrogen and phosphorus removal in each water body. Omitting the open water region of the Solent (which has the highest economic value for nitrogen and phosphorus removal), the four largest estuaries in the SEMS (Portsmouth, Langstone and Chichester Harbors and Southampton Water) have the highest economic value associated with removing nitrogen and phosphorus. This is partly due to the large cleaning capacity of these coastal zones and highlights the potential savings of costs from decreased needs for relatively expensive measures.

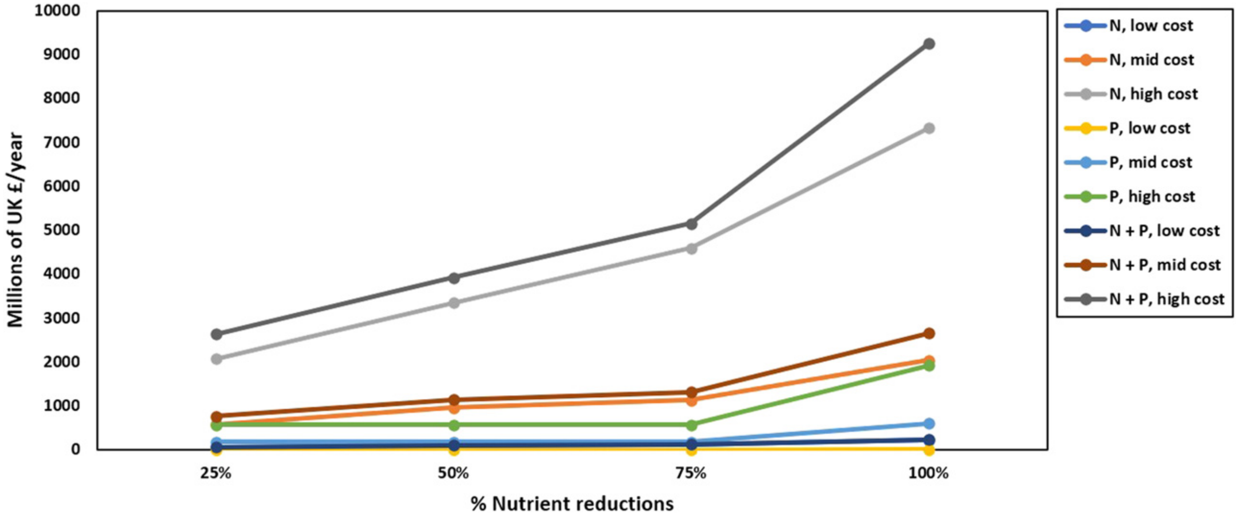

Fig. 5 Replacement value of coastal habitats for nitrogen and phosphorous reductions to the SEMS at low and high marginal nutrient abatement costs.

Table 6 Summary of the estimated nutrient replacement value of habitats in each water body

	Nitrogen (50% Loading)	*Phosphorus (50% Loading)*
Lymington Estuary	£21.30 Million	£5.37 Million
Beaulieu Estuary	£23.56 Million	£4.05 Million
Southampton Water	£86.18 Million	£9.24 Million
Hamble Estuary	£16.52 Million	£8.43 Million
Portsmouth Harbor	£76.22 Million	£30.30 Million
Langstone Harbor	£101.16 Million	£33.93 Million
Chichester Harbor	£157.47 Million	£46.67 Million
Pagham Harbor	£21.15 Million	£3.3 Million
Yar Estuary	£7.86 Million	£2.20 Million
Newton Harbor	£17.74 Million	£7.58 Million
Medina Estuary	£7.11 Million	£2.76 Million
Bembridge Harbor	£2.40 Million	£0.84 Million
Solent Strait (open water)	£423.99 Million	£24.70 Million
Total	£962.65 Million	£179.39 Million

7.5.4.5 Estimated Values of NWR by Different SEMS Habitats

As documented earlier in this chapter Watson *et al.* (2020a) applied a regionally specific replacement cost to value saltmarsh and oysters, this approach was also applied to value littoral sediments, littoral sediments with macroalgae, subtidal sediments, seagrass and reedbeds, with the values provided in **Table 7**. To estimate the economic value of this service a range of regionally sensitive replacement costs are applied, drawing on the locally applicable literature outlined above (Bryan *et al.*, 2013; RSPB, 2013; BPPDC, 2017; River Avon SAC Working Group RAWG, 2019). Average replacement costs of reducing nitrogen and phosphorus from these sources are estimated as 295 [£/kg] for nitrogen and 282 [£/kg] for phosphorus, and these costs are used as mid-range conservative ecosystem replacement value estimates. It should be noted that although the total average value per hectare (£ annualized) for seagrass is negative, if a higher rate of biophysical rate is used to calculate the replacement costs (i.e., representing seagrass in a better condition state) the value can shift to a positive. For example, Watson *et al.* (2022) estimated that if all existing seagrass habitat in the SEMS was improved to 'Good' conditions they would yield a net positive phosphorus removal benefit of £16.47 Million a year.

7.5.5 Management of NWR in Estuarine and Coastal Zones

The sustainable exploitation of benefits provided by the service of NWR depends on our ability to manage waste inputs in relation to the capacity of ecosystems to remediate wastes. Currently, for many coastal zones like the SEMS and Baltic Sea examples above,

Table 7 Summary of the total estimated economic value provided by a hectare of habitat on the UK's Southeast coast. Adapted from Watson *et al.* (2020a) Negative values indicate net loss of the nutrient from the habitat

Unit	*Habitat*	*Biophysical change (median tonnes / year)*	*Total average value per hectare (£ annualized)*	*Total average value (£ annualized)*
Nitrogen	Littoral sediments	827	39,300	243.97 Million
	Littoral sediments (macroalgae)	403	73,578	118.89 Million
	Subtidal sediments	1292	19,559	381.12 Million
	Seagrass	127	53,607	37.41 Million
	Reedbeds	17	18,869	5.15 Million
Phosphorous	Littoral sediments	34	1555	9.65 Million
	Littoral sediments (macroalgae)	479	83,853	135.09 Million
	Subtidal sediments	47	677	13.17 Million
	Seagrass	− 30	− 12,239	− 8.53 Million
	Reedbeds	21	21,448	5.84 Million

the advantages of NWR have turned into threats for the functioning of the coastal zones. The threshold levels, where nutrient and waste loads are cleaned by the coastal zone are being overwhelmed locally and globally well beyond levels that can be sustained under current demands; much less future ones (Beusen *et al.*, 2022). A lack of capacity to manage the service of NWR not only compromises the ability of the marine environment to process our nutrients and waste but also causes a loss of an array of other ecosystem services and benefits for example: food security, raw materials, recreational amenity, sequestration of carbon and an equable environment. Laholm Bay on the Swedish west coast of the Baltic Sea provides an early example where excessive nutrient loads created anoxic sea bottoms that became devoid of plants, fish, and invertebrates in the early 1980s (see Fleischer *et al.*, 1989) leading to potential loss in associated ecosystem services.

In principle, management actions to control nutrient inputs from watersheds to can be divided into two main tasks: (1) identification and choices of adequate measures among the two categories – nutrient reductions to the coast and nutrient harvesting in the coastal zone – for improving NWR and (2) choice and implementation for policies creating suitable institutional frameworks for stakeholders interested in NWR measures. Actions to control nutrient inputs from terrestrial environments are principally essential, so as not to overwhelm a receiving coastal ecosystem's capacity. This can then be followed by adaptive marine regulatory monitoring frameworks necessary to maintain or achieve the objective of 'good' ecological and environmental status respectively in waterbodies and the optional implementation of habitat restoration measures directly into the coastal zones such as cultivation of shellfish, saltmarsh, and seagrass, which can support and enhance the natural capacity of marine systems to mitigate nutrient enrichments. In this section we give a brief presentation of the main nutrient and water-quality management policies in place in the European Union (EU) and UK, and their respective associated experience in achieving the two aforementioned tasks.

7.5.5.1 Water-Quality Management in Europe and the UK

Water-quality management for EU member states is regulated by several directives, either directly with the Urban Wastewater Treatment Directive (UWWTD, 91/271/EC), the Nitrates Directive (91/676/EEC), and the Water Framework Directive (WFD, 2000/60/EC), or within an ecosystem context with the Marine Strategy Framework Directive (MSFD 2008/56/EC). The main and current EU Directive aimed at maintaining the quality of coastal ecosystems is the WFD, while the Habitats Directive (92/43/EEC) (1992) regulates the protection of certain species and habitats. Though the above noted EU Directives no longer apply in the UK, since its withdrawal from the EU, their provisions have however been incorporated into the laws of the UK and its devolved governments through the EU Withdrawal Act 2018 and associated amendments related to EU Exit Regulations, and their principles seem likely to be maintained in the UK (Hughes *et al.*, 2022). Under the WFD, ecological status is assessed in an integrated way through the use of biological quality elements (phytoplankton, benthic flora, benthic invertebrate and fish fauna) together with supporting hydromorphology and physico-chemical parameters, including nutrient conditions (WFD 2000/60/EC). The WFD stipulates that, at good ecological status, nutrient concentrations must "not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of values specified (for good status) for the biological quality elements" (Annex V, 1.2). The quantification and choice of nutrient concentration targets to be achieved are not self-evident, but instead requires EU and UK countries to determine type-specific nutrient criteria ensuring/supporting good ecological status. According to the WFD, a good ecological status of European waters should be achieved in all rivers, lakes, coastal and transitional waters by 2015 or, at the latest, by 2027 (WFD; EC, 2000), However, by the most recent estimate (EEA, 2018; updated with recent data), 40% of EU coastal waters and 66% of transitional waters have failed to achieve this. Equally in UK only 36% of UK coastal water bodies (36%) were in high or good status in 2020 (DEFRA, 2022).

Most countries choose a combination of quantitative targets and appropriate policy instruments to directly and indirectly achieve WFD targets, which in turn usually are implemented in terms of control measures to reduce nutrient and waste reductions to recipient waters. In the UK, the main focus of current control measures are aimed at either reducing nitrogen diffuse pollution from agricultural practices, through Nitrate Vulnerable Zones, the Reduction and Prevention of Agricultural Diffuse Pollution (England) Regulation, various voluntary schemes (e.g., Environmental Land Management scheme, Catchment Sensitive Farming, Environmental Stewardship Schemes, Nutrient Management Plans and the Catchment Based Approach), or to regulate nitrogen and phosphorus pollution from point sources, through permits for discharges from sewage treatment works and industry sites.

The choice of policy control measure is partly determined by historical records, the Polluter Pays Principle (PPP), and also by the localized nature of water-quality management. Although it is relatively straightforward to adopt the PPP by charging those who discharge nutrients at sources, there is a mix of policy choices among the EU member countries. In the UK, the nutrient surplus per unit of land is relatively modest, and the PPP policy design rests mainly on soft instruments such as training and information in order to comply with reducing nitrogen-based fertilizers into Nitrate Vulnerable Zones. This is in contrast with the two countries with the largest nutrient surplus per unit of land in the EU member states: the Netherlands and Denmark. The intensive agriculture in the Netherlands implies a major challenge to follow the Nitrate Directive, and the country has set stringent standards on livestock intensity and quite high PPP fees for nutrient application in excess of a standard. Denmark has also chosen quite strict instruments, where quotas on nitrogen application are set for each farm. Danish legal regulations regarding wastewater discharges are also one of the most restrictive in the EU countries (Preisner *et al.*, 2020) with tax rates regarding the treated wastewater discharged into receiving waters are set for three parameters: BOD_5 (2.47 Euro/kg), Total Nitrogen (4.44 Euro/kg), and Total Phosphorus (24.46 Euro/kg).

However, for measures affecting leaching and diffuse waste inputs, it is not always possible to charge nutrient and wastewater recycling at the spot in question, where challenges have been met by investment in end-of-pipe treatment by water companies, often at significant cost both financial and in terms of greenhouse gas emissions (Meng *et al.*, 2016). Instead, a compensation payment needs to be implemented in order to create the correct incentives. Such a combined charge and subsidy system is currently in use in several countries where fees are often levied on fertilizers, and compensations are paid for measures affecting leaching and retention, such as subsidies for restoring or creating wetlands (see e.g., zu Ermgassen *et al.*, 2021). Such measures are often referred too as "Payments for Ecosystem Services" (PES) which describe a variety of innovative, market-based incentive schemes that reward land managers for maintaining and enhancing environmental benefits ("ecosystem services") such as water quality, flood regulation, climate regulation and certain provisioning and cultural ecosystem services (such as biomass and recreational access). PES schemes involve a willing 'buyer', or beneficiary, of an ecosystem service, voluntarily paying a 'seller' (typically a landowner) who is willing to adopt measures to provide a particular ecosystem service or services. Intermediaries (organizations who act as brokers to coordinate buyers and sellers) and knowledge providers are also important actors in the functioning of PES schemes.

The compensation payments for environmentally friendly land use are to a large extent a result of agri-environmental policies under The Common Agricultural Policy (CAP), which started in the 1980s. Environmental programs affecting agriculture have been incorporated into Pillar 2 of CAP by the cross-compliance requirements, which require minimum environmental standards for becoming eligible for farm payments. These standards focus on both positive and negative externalities and promote, for example, organic farming which is assumed to create less negative and more positive externalities than conventional farming. Most countries implement uniform payments per unit land, for example, for organic farming and wetland construction, which are not differentiated according to environmental impacts. The UK Government is also phasing out CAP-style direct payments and is introducing PES style payments for farmers to provide public goods such as environmental and biodiversity improvements. These changes are taking place during a seven-year 'Agricultural Transition' period running from 2021 to 2028.

7.5.5.2 Nutrient Trading Markets and Nature Based Solutions

In addition to substantial changes in agricultural land use, the increased urbanization of catchments, reflected in new housing has led to significant increases in nutrient loading and reductions in water quality in coastal environments. These impacts have led to a number of regulators in the EU and the UK to stop or slow developments within certain catchments to prevent further deterioration of water quality (e.g., Miller and Hutchins, 2017). Under the Habitats Regulations, 'competent authorities' such as local planning authorities and the Environment Agency must assess the environmental impact of projects and plans (such as planning applications or local plans) which affect habitats sites. Local planning authorities can only approve a project if they are sufficiently certain it will have no negative effect on the site's condition. As a result of these regulations and domestic and European case law, developers must increasingly demonstrate that new projects and developments are 'nutrient neutral'. In response, a number of councils, wildlife trusts, private companies and even the UK regulatory body Natural England have developed land and coastal-use change nature-based nutrient mitigation solutions. This typically involves selling 'nutrient credits' to housebuilders by creating new woodland or wetlands to strip nutrients from water or creating buffer zones along rivers and other watercourses thus mitigating the 'nutrient load' generated by the population growth due to new housing developments. This in turn allows developers to meet their nutrient mitigation obligations and enable local planning authorities to grant planning permission.

The Solent and Teesmouth Nutrient Trading Market Pilots in the UK are two examples of private and government lead nutrient credit mitigation schemes, with a similar option whereby a wildlife trust or Natural England can issue nutrient credits based on the conversion of current intensive agricultural land (e.g., for the commercial production of animals, fruits, crops and/or vegetables) to other land-uses, such as greenfield, woodland, or wetland as part of the schemes (DEFRA, 2023). Currently, the estimated price of a nitrogen credit (equivalent to 1 kg of nitrogen per year for the lifetime of the development, generally 80–125 years) is £1825–3000 (NE, 2020; DEFRA, 2023). While the vast majority of UK offsets included within these schemes are terrestrially based, there is increasing interest in the trading of nutrient credits for the catchment management of nutrients using fully marine-nature based solutions.

In the United States, an early adoption of a marine based nutrient trading system for the Chesapeake Bay, was introduced under voluntary initiatives that took place among the states of Virginia, Maryland, and Pennsylvania (DePiper *et al.*, 2017). As an approach to reducing the costs of meeting the Chesapeake Bay Total Maximum Daily Load (TMDL) the value of nitrogen and phosphorus removal by clams and oysters has been estimated by as a nutrient best management practice, with removal ranging from 26 to 73 kg acre^{-1} year^{-1} (Reichert-Nguyen *et al.*, 2019). A 2018 valuation for the presence of a nutrient credit trading market (nitrogen only) using oyster aquaculture, with high or low credit prices ranged from US$10–$190 per lb of nitrogen (Weber *et al.*, 2018). Recently, the Chesapeake Bay Program Oyster Best Management Practice (BMP) Expert Panel has evaluated and approved nitrogen and phosphorus removal reductions by cultured oysters whereby local development jurisdictions are allowed to use nutrient credits from oyster tissue to count toward fulfillment of nutrient reduction goals (Oyster BMP Expert Panel, 2016). These trading schemes are currently based on harvested tissue only as it is more widely considered (e.g., Rose *et al.*, 2015; Dvarskas *et al.*, 2020) that more research is needed to make recommendations for the development of a nutrient trading schemes that incorporate estimates for nitrogen and phosphorus deposition in oyster shell, denitrification, and burial in sediments.

In addition, the use of wetlands, kelp and other seaweeds are increasingly being cultivated and harvested around the world for the purpose of nutrient removal. Examples of using wetlands as marine-nature based solutions have again largely been pioneered in the United States with several watershed level nutrient trading programs in operation (Shortle, 2013; Saby *et al.*, 2021). The use of kelp and seaweed, however, has gained little traction for its potential role in targeted nutrient credit trading schemes, this is despite several studies from the United States (e.g., Racine *et al.* (2021)), EU (e.g., Hasselström *et al.*, 2020) and China (e.g., Xiao *et al.*, 2017) highlighting its potential value as a nature based solution for removing large quantities of nitrogen and phosphorus from coastal ecosystems. Inertia in credit trading and occurrences of market power has in many instances been combated by the establishment of a third party which is supposed to facilitate credit trade between buyers and sellers. However, thin markets, that is, markets with only a few actors, and lack of trading have been serious obstacles for efficient and fair trading in nutrient credits generated by kelp, seaweed, wetland and bivalve cultivations.

7.5.5.3 Management Actions to Restore Protected Sites to Favorable Condition

Nutrient neutrality can only be an interim solution while we speed up action to tackle nutrient pollution at source and improve the condition of protected coastal and marine sites. The regulation of biogeochemical cycles and ecosystem processes responsible for removing or degrading wastes can protect and enhance the service of NWR. For example, improving the management and condition of wetlands including saltmarsh and seagrass beds can enhance regulatory ecosystem service flows of nutrient and carbon sequestration and storage (Adams *et al.*, 2012; Salinas *et al.*, 2020). It is recognized that marine sediments and biomass are two of the largest storage reservoirs for organic nitrogen and many other wastes in the environment. Furthermore, the ecosystem process of denitrification can help to control the rate of eutrophication, particularly in marine coastal ecosystems subject to large inputs of anthropogenic nitrogen. As such targeted marine restoration actions and nature based-solutions can be developed around these major removal mechanisms. These solutions are based around three main ecosystem processes connected to the service of NWR:

- Assimilation in biogenic material (Sequestration in biomass). Processes that sequester nutrients and other wastes (e.g., metals and plastics) in a habitat or organisms' biomass or other form of biogenic material (e.g., bivalve shells) in such a way that they are not biologically available and do not exhibit toxicity. Essentially sequestration may be reversible if conditions are altered, with the nutrients or wastes returned to harmful forms (Watson *et al.*, 2016). Extraction of these nutrients or wastes can also occur via the harvesting or removal of the organisms (e.g., bivalves, seaweeds or macroalgae) – thereby returning nutrients or wastes back to land.
- Burial in sediments (Long term storage): Although a relatively small sink for nitrogen compared to denitrification (below) the permanent burial of nitrogen, phosphorus and other wastes containing organic compounds (e.g., persistent organic pollutants) is a well-defined final sink for nutrients and wastes in the marine environment. It is important to recognize that nutrients and wastes stored in sediments may not be permanent and may be released if the habitat is disturbed by human pressures (e.g., sediment abrasion from fisheries trawling) or if the habitat extent or condition is eroded. Thus, implementing natural or protected status for coastal NbS that provide NWR will prevent further degradation of ecosystems and prevent nutrient and waste releases into the wider environment.
- Denitrification (Export to another system): This is an important ecosystem process that permanently acts to remove excess nitrogen, through the conversion of nitrate to nitrogen gas, leading to loss of nitrogen to the atmosphere. Denitrification is the dominant nitrogen removal process in wetlands (e.g., saltmarsh, reedbeds) that receive high nitrogen loadings from agricultural runoff or wastewater treatment plant discharge. Enhanced denitrification is also provided by artificially established or re-established bivalve beds, e.g., oyster reefs (Kellogg *et al.*, 2014) or by bivalve aquaculture (Humphries *et al.*, 2016).

NbS focused on protecting and enhancing the three processes above can deliver benefits for both eutrophication mitigation, especially by enhancing NWR and for the waste management of companies and other businesses that derive benefits from the service at the local level. Implementing marine NbS should encompass the protection of existing habitats, the restoration of ecosystems that have been degraded, the sustainable management of working land and marine systems and the creation of novel ecosystems (Riisager-Simonsen *et al.*, 2022). The protection of existing habitats prevents the further release of nutrients and wastes from sediments through land conversion in terrestrial systems, and reduction of seabed activity in the marine realm, safeguarding the biodiversity that depends on such habitats, as well as the wider ecosystem services they provide. The restoration of degraded habitats can actively improve the ability of natural systems to remove nutrients and wastes (e.g., through enhanced denitrification) from the marine environment, as well as recover biodiversity and ecosystem services. Finally, the creation of novel habitats, sometimes called 'gray-green engineering' (Singhvi *et al.*, 2022) on or around near-shore and offshore man-made structures can also help society adapt to the adverse effects of human developments. For example, the potential nutrient and waste mitigation benefits of farming seaweed and bivalves alongside offshore wind arrays is currently unknown but trials and feasibility studies including in the North Sea (van den Burg *et al.*, 2016) could provide a blueprint for how such farms coupled with seafloor conservation measures could be used for larger-scale nutrient bioextraction in the future.

Another future challenge will be to better understand how additional regulatory ecosystem services benefits in monetary terms that would be realized if new NbS priority habitats were created or if existing habitat condition were improved. Current approaches to measure ecosystem services within NbS assessments are generally coarse, often using a single figure for ecosystem services (e.g., nutrient remediation or blue carbon sequestration) applied to the local or national habitat stock, which fails to take account of local ecosystem conditions and regional variability (Watson *et al.*, 2022). Supplies of ecosystem services relating to ecosystem condition have recently been conceptually correlated with WFD ecological status classifications in terrestrial and freshwater ecosystems (Grizzetti *et al.*, 2016; Burkhard *et al.*, 2018). Expanding this framework to better understand the relationship between marine and coastal conditions and services would aid in design measures to protect and enhance the ecological and environmental status of marine ecosystems. A recent study by Watson *et al.* (2022) suggests that incorporating WFD and other comparable marine indicator classifications in localized natural capital accounts could improve regulatory benefit estimates by 11%–67%. This evidence of the potential value of including condition indices in assessments is highly relevant to consider when investing in water ecosystems conservation and restoration as called for by the UN Decade on Ecosystem Restoration (2021–2030), and more generally in global nutrient neutrality policy strategies.

7.5.6 Conclusions

The important role of NWR by coastal zones as an input into the provision of several types of ecosystem services has been known for decades, in particular by natural scientists. In spite of this concern, there are relatively few environmental economic science studies analyzing the particular social values of NWR and appropriate management and policy instruments to combat the threat to NWR. This is not surprising, given the different questions addressed by natural and social scientists. Despite the relative lack of study, this chapter addressed two main questions: What are the economic values associated with the ecosystem service of NWR? And, what characterizes good management of this service? With respect to the first question posed in the chapter – the social value of NWR by coastal waters – it was found that there are very few realized studies estimating such values (i.e., studies based on market values). Instead, valuations have been made of potential nutrient remediation value using replacement, abatement, and other forms of indirect costs when no one is paying for them (i.e., no credits are purchased or traded). Such values use sequestration of other land and sea uses, notably wetlands or shellfish, and of the effects of NWR by coastal habitats, to estimate mitigation of eutrophication of marine waters. The results from these studies point at significant contributions of nutrient and waste sequestration by wetlands which can range between €0–172,227 using a replacement cost valuation approach.

Due to the lack of realized studies estimating NWR by coastal waters, the chapter presents novel results from calculations of potential NWR values for the Baltic Sea and for the SEMS. The results indicate average replacement costs for nitrogen & phosphorus varies between 9 million Euros/region/year and 1.1 billion Euros/region/year, with nitrogen removal from the water column a more economically valuable service compared to drawdown of phosphorus and carbon (see Watson *et al.*, 2020a). The high value is attributed to the proportionally high uptake of nitrogen compared to phosphorus, the high monetary value allocated to nitrogen removal, and the fact that nitrogen does not need to be transported to the sediments to be effectively removed, due to the export removal process of denitrification. While there are inherent mechanistic differences between coastal and fully marine or ocean-based removal, these equivalencies are necessary in the absence of market-based values for these processes (Costanza *et al.*, 2017) and help contextualize potential values for the service of NWR.

When investigating policy measures and management actions for combating threats for degradation of NWR by coastal waters, two main issues were identified, namely, the choice of policy instruments and the use of management measures to achieve NWR ecosystem service enhancements. With respect to the choose of measures that may improve NWR, the scientific literature points to the potential of adapting measures to uses already ongoing in coastal waters, such as bivalve and kelp farming, as well as restoring structured marine habitats such as saltmarsh, seagrass and oyster reefs which can provide NbS for nutrient management through assimilation, burial and sediment denitrification (Hughes *et al.*, 2022; Watson *et al.*, 2020a). These measures have not yet been generally recognized in actual policymaking. Two exceptions are the trading in nitrogen credits allowed between housing developers and local public bodies in the UK to mitigate nitrogen enrichment and or the purchase of nutrient credits from aquaculture-site mitigations in the USA. Although considerable progress has been made in these pilots, the ultimate success of some programs (e.g., the emergence of nutrient trading markets in the Chesapeake Bay drainage) has yet to be proven. A barrier to scaling up nutrient trading schemes using NbS is the availability of finance, marginal costs, and supply and demand (Thompson *et al.*, 2023). Sources to overcome these barriers include philanthropy, voluntary nutrient offsetting, public sector grants, agri-environment schemes, PES (such as water companies paying farmers to reduce pollution runoff), or regulatory obligations for firms to offset the impacts of their actions, such as biodiversity offset markets arising from net gain requirements. Land and sea owners need incentives to cover the opportunity costs of restoration activities, as well as direct investment costs. NbS can be made more investable by stacking multiple benefits for NWR, biodiversity and other services such as flood protection or carbon sequestration and storage, financed by different beneficiaries and supported by a greater range of multi-sectoral legislative or policy instruments.

It was also recognized that actual policymaking in EU and the UK, while similar with regard to compliance with the WDF and MSFD, are increasingly divergent with respect to both choice and design of nutrient mitigation instruments. Policies in the EU are focused on the sources and causes of positive and negative externalities, such as the promotion of organic farming and wetland construction, which are supposed to decrease negative and increase positive externalities. In the UK, policies are more directed toward the negative externalities and outcomes, such as nutrient and waste effluents associated with housing developments. Nevertheless, policies in the EU and UK directed toward nutrient runoff from both point and nonpoint source have resulted in decreased nutrient loads to coastal waters in many countries (Vigiak *et al.*, 2023). However, additional factors in the future may make NWR by coastal ecosystems more difficult. Climate change is one such factor; if rising seawater levels and increased storm surges damage or destroy coastal wetlands, for example, their ability to mitigate nutrient loads may be compromised. Furthermore, some evidence exists of recipient ecosystem hysteresis, that is, habitat or species facilitated nutrient abatement sometimes fails to reduce eutrophic conditions, even though nutrient loads go down (Duarte *et al.*, 2009). These causes are unclear but suggest that complicated and poorly understood feedback may result in continued poor water quality. Hence, future study of the effectiveness and economics of coastal NWR may also require increased understanding of the abiotic component of the service including coastal ecosystem dynamics and exchanges with marine waters. However, regardless of the causes of coastal ecosystem degradation, further knowledge improvements need to be made with respect to the links between the status of the ecosystem and its provision of ecosystem services. Today, it is not even settled which indicators are most useful for expressing habitat extent and condition relating to improvements in NWR and water quality, although Biodiversity Quality Elements associated with the WFD have been suggested (see Watson *et al.*, 2022). There is thus an urgent need for identifying and quantifying the causal relations between appropriate indicators of NWR and also its links to the provision of other ecosystem services such as the provision of fish as food, biodiversity, and recreational values.

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Appendix A

See **Table A1**.

Table A1 Minimum cost with and without NWR for the BSAP targets, population and GDP/capita in 2006 from Gren *et al.* (2013)

Country	*Minimum costs without NWR*	*Minimum cost with NWR*	*Population, 1000*	*GDP/capita, 1000 Euro*[3]
Denmark	95	72	4600	35
Finland	199	78	5261	27
Germany	13	4	3300	24
Poland	1751	1150	38,140	6
Sweden	162	101	9078	29
Estonia	137	55	1340	8
Latvia	255	203	2289	6
Lithuania	314	250	3409	6
Russia	244	145	8878	5
Total	3171	2058	76,294	12

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