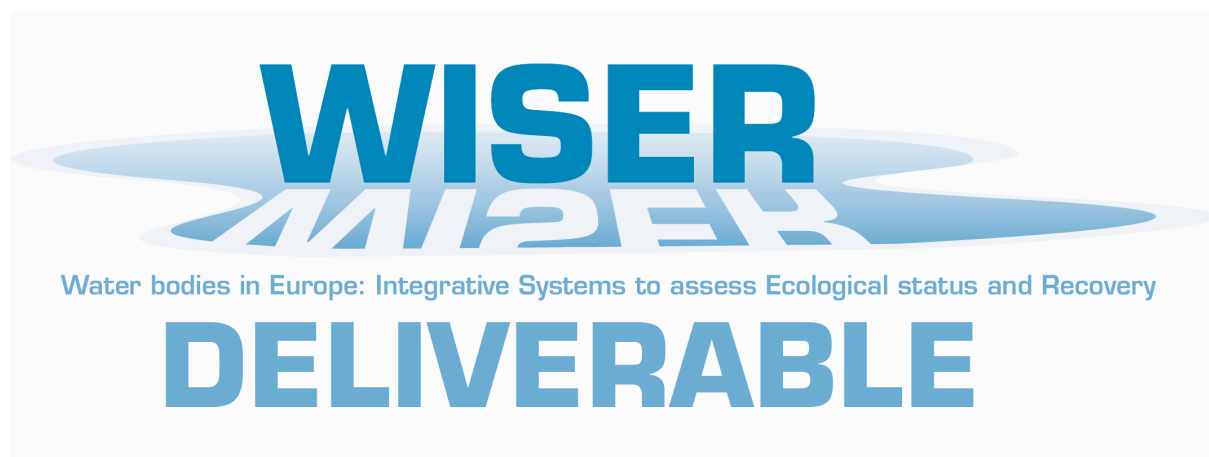Collaborative Project (large-scale integrating project)
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WISER

Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery

DELIVERABLE

Deliverable D5.3-5: Effects of nutrient reductions in transitional and coastal waters — Synthesis and Guidelines

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Content

SYNTHESIS

Translation of results to specifically address the practitioners

Temperature effects on hypoxia and benthic fauna

Hypoxia is a mounting problem affecting the world's coastal waters, with severe consequences for marine life, including death and catastrophic changes. The deleterious effects of hypoxia are amplified by warming. Global warming will contribute to decrease the global average dissolved oxygen in the oceans worldwide, and will also affect the oxygen requirements of marine benthic macrofauna. Increasing temperature diminishes oxygen solubility and increases the respiration rates of organisms, as temperature plays a fundamental role in regulating metabolic processes. Ocean warming increases the vulnerability of benthic macrofauna to reduced oxygen, increasing the mortality of benthic fauna and greatly extending the area of coastal ecosystems affected by hypoxia-driven mortality. For more details, see WISER deliverable D5.3-1 and Vaquer-Sunyer & Duarte (2011).

Hydrogen sulphide exacerbates effects of hypoxia

Hypoxia is a mounting problem affecting the world's coastal waters, with severe consequences for marine life. Hypoxia is emerging as a major threat to marine coastal biota. The deleterious effects of hypoxia can be amplified by the presence of hydrogen sulphide. When the oxygen content of the waters is depleted and the oxic layer of the sediment migrates upward to the water layer, sulphide is also moved from the sediment to the water and become another stressor to benthic organisms that try to cope with hypoxia. For more details, see Vaquer-Sunyer & Duarte (2010).

Phytoplankton biomass yield relative to nutrients have doubled

Nutrient management plans in Europe and North America have apparently been unsuccessful in combating eutrophication. Many of these plans were based on relationships of increasing nutrient levels versus chlorophyll measured during the eutrophication phase, and rested on the assumption that reducing the inputs of nutrients and consequently nutrient level would similarly lead to a reduction in phytoplankton biomass. Thus, it was assumed that if nutrients were reduced to a lower level of the past then the corresponding levels of chlorophyll should also be achieved. This fundamental tenet is flawed because other pressures such as temperature increase, acidification, overfishing etc. have shifted the baselines and potentially induced a regime shift. For more details, see WISER deliverable D5.3-1 and Carstensen et al. (2011).

Hypoxia makes ecosystem recovery more difficult

Coastal hypoxia is increasing in the global coastal zone, where it is recognized as a major threat to biota. Hypoxia is defined as oxygen concentrations below a certain value, typically 2 ml/l or 2 mg/l, but the deleterious effects on the ecosystem already start at higher oxygen concentrations. Knowing the thresholds that fundamentally lead to a change in ecosystem functioning is important to quantify for management. Moreover, these thresholds are not static but regulated by other processes, associated with both local and global pressures on the system, particularly warming. Exceeding the critical thresholds associated with hypoxia may require even further

nutrient reductions to restore a well-functioning benthic community. However, recovery from hypoxia is possible. For more details, see Steckbauer et al. (2011).

Recovery of estuarine and coastal ecosystems takes decades

Anthropogenic pressures alter natural ecosystems and the ecosystems are not considered to have recovered unless secondary succession has returned the ecosystem to the pre-existing condition or state. However, depending upon the scales of time, space and intensity of anthropogenic disturbance, return along the historic trajectory of the ecosystem may: (i) follow natural restoration though secondary succession; (ii) be re-directed through ecological restoration, or (iii) be unattainable. From a review of long and medium-term monitoring networks we did, we show that although in some cases recovery can take <5 years, especially for the short-lived and high turnover biological components (i.e. plankton, meiofauna, etc.), full recovery of coastal marine and estuarine ecosystems from over a century of degradation can take a minimum of 15-25 years for attainment of the original biotic composition and diversity may lag far beyond that period (i.e. in seagrasses, hard-bottom substrata macroalgae, etc.). For more details, see Borja et al. (2011).

Loss of benthic vegetation sustains a turbid regime

Seagrasses constitute an important biotope in coastal ecosystems, but there has been a global decline over the last century. This is a consequence of eutrophication stimulating growth of plankton and thereby reducing the light penetrating to the bottom. In response to this the depth limit of seagrasses, in temperate waters typically eelgrass, has decreased. Relationships linking nutrient levels with eelgrass depth limits, established on data during the eutrophication phase, have been proposed as nutrient management tools. However, such relationships are not valid for predicting the response to decreasing nutrient levels due to shifting baselines and feed-back mechanisms, where lack of benthic vegetation increases resuspension of sediments and thereby maintains a stable turbid regime. For more details, see WISER deliverable D5.3-4.

Conceptual framework of potential effects of degradation and management

Transitional and coastal waters are not isolated entities that can be easily separated from the surroundings and where drivers-pressures-state-impact-response-recovery chains can be developed for specific drivers and pressures. Transitional waters are today affected by multiple pressures (Fig. 1), including nutrient enrichment from the watershed as well as from the open boundary, climate change, overfishing, contaminants, invasive species, aquaculture and physical alteration. Although the latter is not a common pressure, as opposed to the other pressures that are observed in almost all European water bodies, there are specific sites, most pronounced for transitional waters, where physical modifications significantly affect the ecological status.
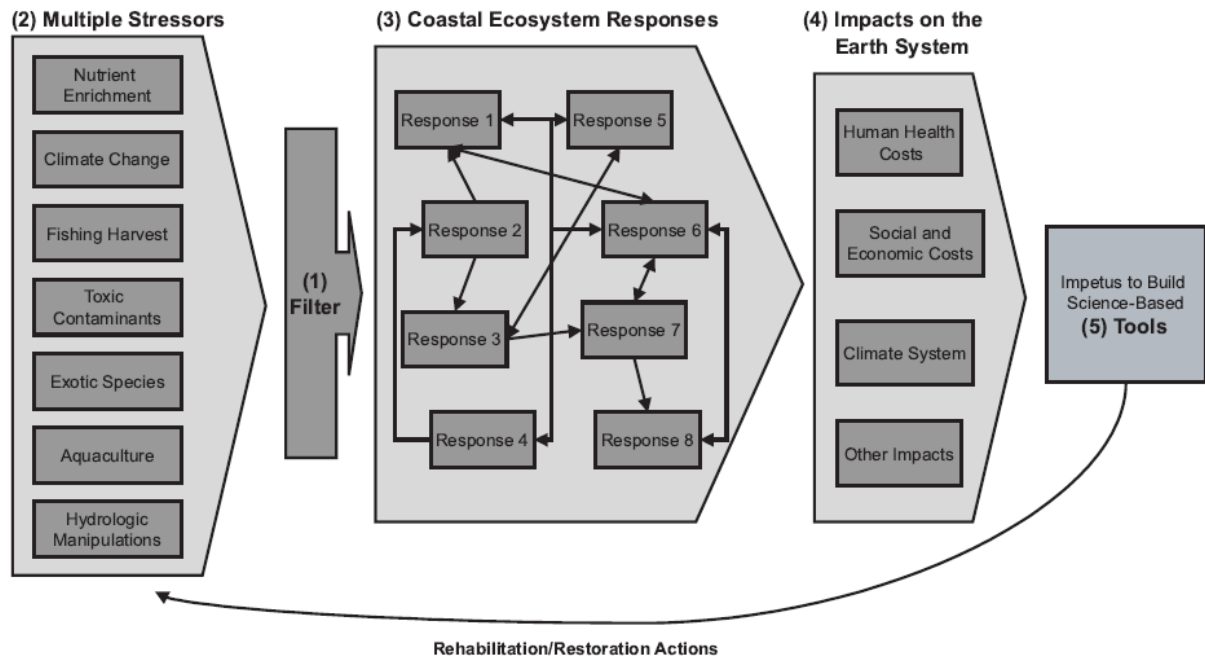
Figure 1: Conceptual model showing the multiple pressures affecting various ecological components in transitional and coastal waters and that these stressors are modulated by system-specific characteristics such as tidal mixing and retention times. These changes affect the ecosystem services and ultimately services directly relevant to humans such as economy, health, climate, etc. These consequences should give impetus to develop science-based tools for appropriate management of stressors in estuarine and coastal waters. Conceptual model from Cloern (2001).

Water bodies have specific characteristics regulating the sensitivity of the ecosystem to specific pressures. These characteristics are mostly of physical nature such as hydrological residence times, stratification patterns, tidal mixing, wind mixing, bathymetry, sediment and substrate types, exchanges with the open boundary, salinity and temperature, and so forth. For example, an estuary with tidal mixing is less vulnerable to hypoxia than one without any significant tidal mixing.

In the Water Framework Directive (WFD) the fundamental tenet has been that aquatic systems are generally bottom-up controlled, but over the last decade there is increasing evidence that top-down control and consequently food-web alterations may be equally important in many transitional and coastal waters. For example, in the Gulf of Riga phytoplankton biomass has increased over the last couple of decades despite that nutrient levels have declined, but Jurgensone et al. (2011) showed that this could be explained by reduced zooplankton grazing pressure most likely associated with overfishing in the Baltic Sea. Thus, the complex mosaic of interacting biogeochemical and biological processes at all trophic levels clearly suggests that individual components and effects cannot be isolated, but the transitional and coastal ecosystems must be considered in their entirety. Fortunately, this complexity is acknowledged to larger extent in the Marine Strategy Framework Directive (MSFD).

Actual effects of degradation and management

It has been a pervasive belief underlying the WFD that degradation and recovery processes would follow the same pathway, such that reducing nutrient inputs and concentrations to a level

of the past would return the ecosystem into the corresponding ecological status. A bulk of studies has accumulated documenting that this fundamental tenet is flawed. Duarte et al. (2009) clearly showed that phytoplankton biomass did not return to a past level after nutrient input reductions in 4 coastal ecosystems, and introduced the concepts of shifting baselines and regime shifts in the marine scientific literature. Carstensen et al. (2011) followed up on this by clearly demonstrating that the chlorophyll yield per unit of nitrogen has almost doubled over the last 3-4 decades in 28 different coastal ecosystems and discussed that these changes are the likely consequences of climate change, overfishing and other factors associated with human over-exploitation of the planet.

Hypoxia associated with organic material loading is one of the most deleterious effects of eutrophication, because it fundamentally alters ecosystem function by recycling a large portion of nutrients back to the water column from the sediments (Steckbauer et al. 2011). There are only few studies documenting recovery from hypoxia following nutrient reductions as opposed to many studies reporting lack of recovery. The sites that have recovered from hypoxia have experienced considerable reductions in nutrient inputs, typically from point source abatement, and recovery did not come instantaneously but took decades. The recovery pattern can, to some extent, be almost as abrupt as the degradation phase, and it has been shown that recolonisation of bioirrigating benthic macrofaunal species can lead to a faster recovery pattern, because of improved nutrient removal in the ecosystem itself (Norkko et al. 2011). A delayed response was also observed in the Nervion Estuary in Spain, where benthic communities and fish recovered 10-15 years after the onset of wastewater treatment, mostly because metals and organic pollution from the past were only slowly reduced in the sediments (Borja et al. 2010). Within this estuary, biodiversity (including plankton, macroalgae, macrofauna and fish) value increases after abatement of nutrients and increase of oxygen (Pascual et al., 2012). Additionally, in some semi-enclosed Baltic estuaries the recovery from pollution has been delayed due to benthic release of nutrients even under good near-bottom oxygen conditions and without direct nutrient loading (see WISER deliverable D5.3-3).

Hypoxia is one of the most adverse effects of eutrophication that is expected to increase in the future due to reduced solubility of oxygen in water and increased respiration with increasing temperatures. Some transitional waters may even switch from net autotrophy to net heterotrophy with warming conditions. Moreover, due to increasing respiratory demands with temperature increases the sensitivity of various organisms to hypoxia will also change and critical oxygen thresholds will be higher in the future (Vaquer-Sunyer & Duarte 2011). Another confounding factor is the presence of hydrogen sulphide, which is toxic to most organisms and accelerates mortality (Vaquer-Sunyer & Duarte 2010). Sulphide is always present in the anoxic layer of the sediments, but in an organically enriched sediment the oxic sediment layer can be quite thin, despite reasonable oxygen conditions in the water column, essentially leading to a benthic habitat compression and increased sensitivity of benthic species to fluctuations in oxygen levels, because the sulphide front is just below the sediment surface. Reduced nutrient inputs will lead to a reduced organic loading of the sediments, but it may take decades before the organic pool stored in the sediments has been respired or permanently buried and consequently, the sulphide

front has been brought deep down into the sediments to allow a resilient benthic community to establish.

Nutrient enrichment during the eutrophication phase led to a gradual outshading of benthic vegetation over a couple of decades, and it was anticipated that reduced nutrient levels would lead to a fast recovery and re-establishment of benthic vegetation. This has, however, not been the case since reduced nutrient inputs has primarily reduced the dissolved inorganic fraction of the nutrient pool in the water, which does not attenuate light at those wavelengths used for photosynthesis. As a consequence of this inconsistent reduction in the different nutrient fractions the light regimes that eelgrass and other benthic plants experience is basically unaltered. Thus, removal of benthic vegetation from large areas of shallow marine waters fundamentally alters the habitat and makes recovery difficult. An increased resuspension of silt and clay particles has been observed after loss of benthic vegetation, which also attenuates light and thereby compensates for the small reductions in organic matter in the water column.

Indicators of degradation and management

A good indicator signals changes in various stressors, is ecologically meaningful and easy to monitor. However, there can be large differences in the complexity and time responses of indicators to changing stressor levels. Despite decreasing nutrient levels in many regions around Europe recovery of all ecosystem components has not been observed yet. This lack of clear signs of recovery has led to frustrations amongst managers and has even led people to suggest abandoning of indicators, if responses to reduced nutrient levels were not simple and direct. The WFD has dictated the use of four biological quality elements in transitional and coastal waters and no indicator for any of the four biological quality elements respond to e.g. nutrient status in a simple and direct manner. Although nutrient concentrations in transitional and coastal waters mostly have a direct linear response to nutrient inputs (Carstensen & Henriksen 2009), the combination of top-down versus bottom-up control in the food-web and cascading effects across the different trophic levels suggest much more complex responses for the biological components (Fig. 1).

Phytoplankton biomass, typically measured as chlorophyll a, is believed to be a good indicator of eutrophication, because many studies reported increases during the eutrophication phase and bioassays documented simple and proportional responses to the addition of the limiting nutrient. However, this simplicity has been challenged during the oligotrophication phase when most time trajectories of marine ecosystems having experienced both eutrophication and oligotrophication do not exhibit the same simplicity of response in the latter phase as was observed in the first phase (Carstensen et al. 2011). This could be interpreted as chlorophyll being inadequate as indicator of oligotrophication. However, it is unquestionable that nutrients stimulate algae production as a direct causal link, whereas the algal loss processes can be more difficult to quantify. Scheffer et al. (2005) conceptually showed how overfishing could lead to reduced pelagic grazing through trophic cascades, and Petersen et al. (2008) and Cloern et al. (2007) have shown that the presence of benthic filter feeders can change chlorophyll levels significantly. Thus, phytoplankton biomass is an ecological relevant indicator, but to link it to

the nutrient pressure loss processes must also be accounted for as shown in Jurgensone et al. (2011).

Indicators from the benthic communities (both vegetation and fauna) are affected by changes in the phytoplankton community through outshading and as a food source in addition to the risk of hypoxia from enhanced sedimentation of organic material. As such, the responses of benthic indicators to nutrient status can be just as masked as the phytoplankton response. Moreover, sediment characteristics, typically a cumulative effect of the eutrophication and oligotrophication phases, has an important effect on benthic indicators (Krause-Jensen et al. 2011) leading to delayed and threshold responses of recovery.

Ecological and environmental thresholds

In the Gulf of Riga, Jurgensone et al. (2011) found that phytoplankton biomass was apparently regulated by nutrient levels in spring and zooplankton biomass in summer, suggesting a seasonal shift from a bottom-up to a top-down controlled system. However, the empirical basis did not provide evidence for a threshold response of phytoplankton biomass. On the other hand, the phytoplankton composition did show a threshold response to temperature, suggesting that dinoflagellates were increasingly substituted by chlorophytes once summer water temperature exceeded 16 °C (Jurgensone et al. 2011). It has also been proposed that cyanobacteria should become more dominant with increasing temperatures due to more optimal growth rates with temperatures exceeding ca. 20 °C (Paerl & Huismann 2008).

Eelgrass depth limits are strongly correlated to Secchi depth, clearly documenting that light is the single most important factor. However, sediment characteristics do also play an important role showing threshold responses to various measurements of organically enriched sediments (Krause-Jensen et al. 2011). In more porous and organically enriched sediment with high water content eelgrass need more root anchoring to establish and this requires more light to compensate for a larger leave/root biomass ratio. Thus, the light compensation point is higher once the organic content of the sediments exceeds a threshold value. This threshold response could be another explanation to the problems of eelgrass recovery observed in many European regions.

The threshold of hypoxia was originally defined as oxygen concentrations below 2 ml/l (Diaz & Rosenberg 1995), a value that was often perceived as a universal threshold of no effect of low oxygen versus detrimental effect of low oxygen. However, recent studies have shown that sublethal effects of low oxygen start already at 4-5 mg/l and vary greatly across marine benthic organisms (Vaquer-Sunyer & Duarte 2008). Furthermore, these critical thresholds for oxygen also vary with temperature (Vaquer-Sunyer & Duarte 2011) and the presence of hydrogen sulphide (Vaquer-Sunyer & Duarte 2010).

Effect of climate change (temperature) and global change

Climate change and the increased demand on world's natural resources pose a major threat to all transitional and coastal ecosystems. The consistent and increasing trends of chlorophyll yield relative to total nitrogen across 28 ecosystems in four regions reported in Carstensen et al.

(2011) most likely result from large-scale processes, possibly associated with global climate change and increasing human stress on coastal ecosystems. Increasing CO_2 concentrations and water temperature, and decreasing pH may alter phytoplankton communities directly at many levels, from physiological processes to nutrient requirements and community structure. Large-scale changes in the food web structure and function of coastal ecosystems derived from overfishing and excess harvest of filter feeders, both global phenomena, erode ecosystem buffers and triggers increased phytoplankton biomass enhancing the vulnerability of coastal ecosystems to eutrophication. Eutrophication and overfishing may lead to proliferation of jellyfish altering the food-web through increasing predation on protists and thus reducing the pelagic grazing pressure on phytoplankton. Thus, the parallel trend toward increase in Chla yield per unit nitrogen over the past decade in all regions could be the result of the major shift in the baselines for the functioning of coastal ecosystems resulting from the combined effects of climate change, overfishing, and, possibly, other components of global change.

Global change has affected basically all components of marine ecosystems over the last 50-100 years (Duarte et al. 2009) - in many cases with resources being exploited beyond the sustainable level (Rockström et al. 2009). The concept underlying WFD, aiming at returning transitional and coastal ecosystems to a previous state is unattainable and a new paradigm should be developed aiming at delivering marine ecosystems to future generations maintaining functional integrity and services of ecosystems conducive to a sustainable future.

BEST PRACTICE MANAGEMENT GUIDANCE

Implications for practical ecosystem management

Shifting baselines must be considered when targets (good-moderate boundary in WFD) for ecosystem restoration are developed; otherwise, it is unlikely that the targets will ever be achieved. The consistent and increasing trend in chlorophyll yield relative to total nitrogen (Carstensen et al. 2011) is a clear indication that other pressures, associated with global change, shift the baseline. It could be argued that additional local measures should be implemented to compensate for the trans-boundary effects associated with global change, but still there is a large risk that local measures in the watershed are insufficient to achieve a target that is defined as a previous state. Therefore, the alternative is to revise the targets in the face of shifting baselines. This concept is illustrated in Fig. 2, which considers what would be the trajectory of a hypothetical coastal area under two scenarios: (1) a "do nothing" scenario, where nutrient concentrations do not change over time and where the ecosystem will exhibit a trajectory that departs from the general regression line to occupy a position of increased Chla due to the increase in the yield of chlorophyll per unit nitrogen over time, and (2) a "nutrient reduction" scenario, where the ecosystem will follow a trajectory that leads to an increase in Chla relative to the initial state, due to the increase in the yield of chlorophyll per unit nitrogen, but that in fact represents a reduction in the realised Chla concentration relative to the "do nothing" scenario (Fig. 2).
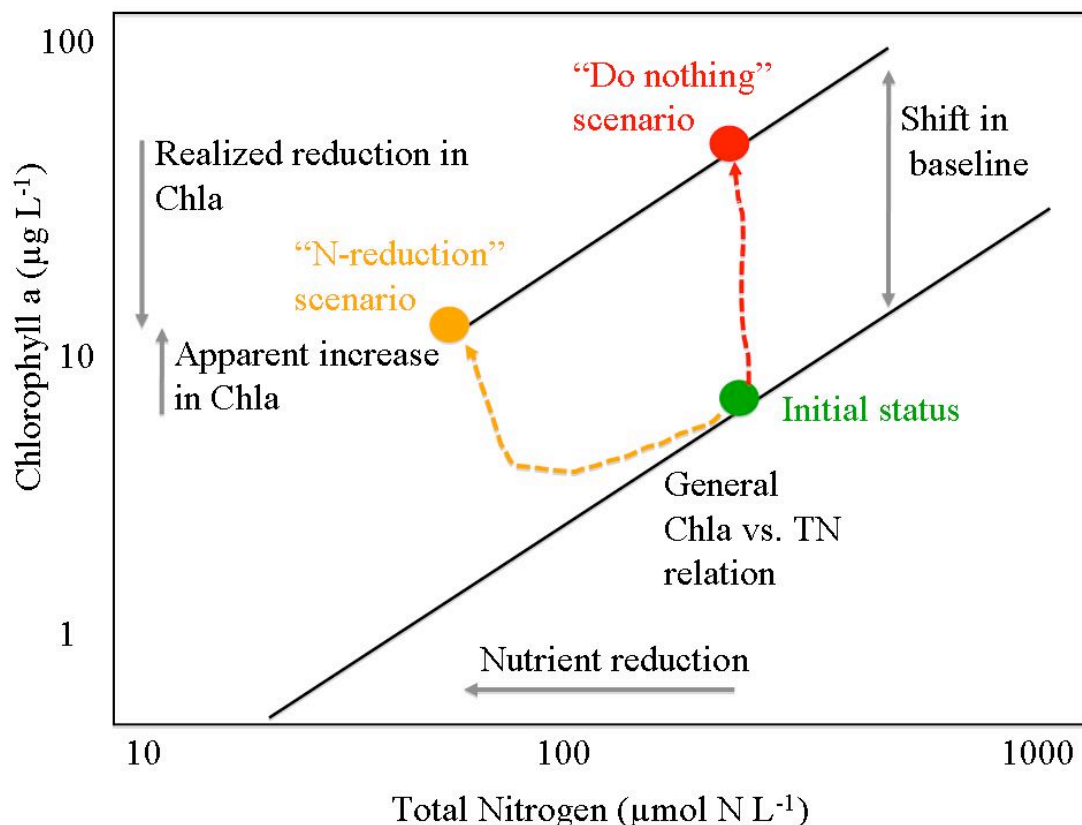

Figure 2: Conceptual model demonstrating the implications of shifts in the yield of Chlorophyll a per unit total nitrogen in coastal ecosystems for the evaluation of the outcome of managerial actions to reduce nitrogen concentrations. The possible outcomes of two alternative strategies, "do nothing" and "nutrient reduction", are shown. From Carstensen et al. (2011).

The key message underlying this conceptual model is that shifting baselines imply that the present state of the system is not an adequate reference to evaluate the effectiveness of nutrient reduction plans, as the future status of the ecosystem would differ from that at present under a "do nothing" scenario. Moreover, the confidence of Chla predictions for nutrient reduction scenarios will weaken, as the uncertainty of predicting the shifting baselines has to be included as well. Trajectories of recovery are even more complex in the potential presence of hysteresis responses, and declining Chla could result from small changes in nutrient inputs if a proper functioning of the coastal ecosystem is re-established. Thus, despite observed increases in Chla concentrations it is still important to stress that nutrient reductions do release pressure on the ecosystem and improve conditions relative to what these would have been under a "do nothing" scenario.

Provided the importance of changing baselines for the setting and evaluation of actions to reverse eutrophication, it is fundamental that our understanding of the causes of such shift in baselines improves to allow forecasting the trajectories of individual coastal ecosystems. A better understanding of the dynamics of coastal ecosystems forced by both changes in nutrient inputs, derived from factors operating at the basin scale, and shifting baselines derived from forces operating at various scales is fundamental to achieve this goal. This requires a research agenda that faces the complex interactions, operating at multiple levels, controlling the variability in the yield of chlorophyll per unit nitrogen over time. Developing such tools will be a major aim in setting more realistic targets for ecosystem management.

Practical measures for ecosystem management

Since focus in WP5.3 is on eutrophication, the main management aim is to reduce nutrient inputs to the water body. Nutrient sources derive from the watershed, from the airshed and from exchanges across the open water boundary. The magnitude of the different sources may vary considerably across water bodies, and the cost-effectiveness of different measures to reduce inputs can also be quite variable. Essentially, this may also have geopolitical aspects when nutrient inputs over the open boundary and from atmospheric deposition are significant and reducing local inputs alone will not be sufficient to achieve the targets. In such cases, ecosystem management at the regional level is required. The WFD is essentially not suited for the situation when trans-boundary inputs are large and local river-basin management plans will not do alone. The MSFD is more suited for this situation, when management plans have to be made for the regional seas (Borja et al. 2010).

If nutrient inputs are dominated by local sources then there are many measures that can be implemented to reduce nutrient inputs from diffuse and point sources. Point source reduction can be achieved by improving wastewater treatment, particularly tertiary treatment or biological nutrient removal plants, which also removes large portions of the inorganic nutrients delivered with the wastewater. Another measure is improving the retention capacity of combined sewers such that overflows during urban storms are minimised. Measures to reduce nutrient inputs from diffuse sources are more complex and can be implemented at various scales. One of the most effective measures is to reduce the fertiliser application on arable land, but this also reduces the

crop yield. Use of catch crops during winter and spring is another measure that reduces the leakage of nutrients from soils. Tonitto el al. (2005) demonstrated with a metanalysis the potential for diversified rotations using N- and non-N-fixing cover crops to maintain crop yields while reducing the anthropogenic contributions to reactive N fluxes, so using biological N fixation, could maintain yields while reducing N losses. Reducing livestock densities is another option. Finally, establishing or restoring wetlands within the catchment is another measure to retain nutrients before discharge to the sea. It is naturally important to select the most cost-efficient measures first, and cost optimisation, using various cost functions for the different measures, is an invaluable tool for decision makers.

Key messages for policy makers

- European transitional and coastal waters are affected by multiple stressors, and management plans should address as many stressors as possible in a combined manner. Still, some stressors (e.g. climate change) cannot be fully reverted and will lead to shifting baselines that must be accounted for when setting targets for ecosystem restoration.

- Marine ecosystems display threshold responses to various pressures. The most pronounced thresholds are hypoxia together with the flux of labile organic matter (Lehtoranta et al. 2008) which fundamentally changes ecosystem functioning by reducing the natural capacity of the ecosystem to remove nutrients such that more nutrients are released from the sediments to the water column. This can maintain the ecosystem in an undesirable self-sustaining state. Loss of benthic vegetation similarly leads to an undesirable turbid state that is maintained through resuspension of sediments. Benthic vegetation stimulates the sedimentation of particulate matter and thus maintains a desirable clear water state. Thus, both hypoxia and loss of benthic vegetation can lead to regime shifts with alternative stable states.

- From a management perspective such thresholds leading to sudden shifts are important to quantify, i.e. for defining safe margins of operation in a healthy state and for defining targets for re-establishing a healthy state, if the ecosystem is in an undesirable state. However, the thresholds values change with climate change and variations in other pressures, which calls for a continuous valuation of threshold values and subsequent management.

- Response times to management efforts can be quite long, typically on the order of decades. For example, nutrients continue to leak from sediments long time after nutrient inputs have been reduced, and the benthic community cannot establish before the sediments have obtained a quality allowing re-colonisation of benthic species. This implies that measures implemented in one WFD cycle may not have an effect on the marine ecosystem even in the following cycles.

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