21st century fisheries management: A spatiotemporally explicit tariff-based approach combining multiple drivers and incentivising responsible fishing.

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Abstract

Traditionally fisheries management has focused on biomass and mortality, expressed annually and across potentially large spatial management units. The Ecosystem Approach requires more indicators at finer scales. Quotas and effort control are largely inappropriate for ecosystem drivers, but remain our principle tools. Incorporating ecosystem targets (e.g. habitat damage, mammal bycatch, biodiversity impacts) would need additional and potentially conflicting management tools. This would further complicate an already confusing micromanagement approach which benefits neither industry, nor science, nor policymakers. Here we present a simple management approach that provides incentives for “good behaviour”. Fishers would be given a number of credits, here called RTIs (Real-Time Incentives), to spend according to spatiotemporally varying tariffs per fishing day. Tariffs could be based on fish stock and ecosystem targets. The fisher could choose how to spend his RTIs, e.g. by limited fishing in sensitive areas, or by fishing much longer in less sensitive areas. The data used to set the tariffs would be transparent to assist in that choice. This RTI system does not prescribe and forbid, but instead allows fishers to fish wherever and whenever they want; ecosystem costs are internalized and have to be taken into account by fishers in their business decisions. We envisage the approach to be stand-alone for the fleets operating under the scheme, with no need for catch/landings quota. The approach could facilitate further devolution of responsibility to industry and matches well with industry-developed sustainable fishing plans proposed in the Common Fisheries Policy (CFP). The development of these fishing plans and the enhanced focus on spatial issues would provide greater opportunities for incorporating fishers’ local knowledge and allow for more adaptive and robust management.

Keywords: discards, Ecosystem Approach to Fisheries Management (EAFM), fisheries management, incentives, internalised costs, real-time information, spatiotemporal flexibility, tariffs.
Introduction

In this paper we explore a novel approach to the management of fisheries, (i) that allows the inclusion of multiple and diverse drivers, (ii) that is highly resolved both spatially and temporally, (iii) that can be simply understood at the level of a single vessel’s operational choices, and (iv) that provides incentives for “good behaviour”. It should also move some way towards a reversal of the burden of proof, a results-based management approach and a more devolved and less top-down management.

We live in a changing world, but fisheries management has largely remained static in approach, based mainly on managing the abundance and fishing mortality of single commercial species. This is generally by means of single-species Total Allowable Catches (TACs) along with effort and capacity controls. But these tools are probably inadequate for an Ecosystem Approach to Fishery Management (EAFM), where we would have to include multispecies interactions, and effects on the wider ecosystem. So, presumably, we would then need additional rules and regulations to encompass these wider EAFM drivers, which would have the effect of increasing management complexity.

These ongoing challenges in moving towards EAFM have driven a paradigm shift from top-down regulation (Daw and Gray, 2005) towards strengthened stakeholder participation in fisheries governance, reflected in the EC’s Green Paper on the new CFP (Anonymous, 2009b) and the EC’s proposal for the new CFP (Anonymous, 2011). The EC Green Paper essentially proposes devolution towards more regional-based management and to include stakeholders directly in developing fisheries management plans. Current thinking is that fisheries management plans that include fishers in the decision-making process are more likely to be successful in achieving objectives (Kelly et al., 2006; Kraak, 2011). Alongside these significant changes in the traditional fisheries management process, there are rapid and ongoing developments in technology that can be directly utilised for fisheries assessment and management. For example, real-time high-resolution spatiotemporal data on vessel activity [Vessel Monitoring Systems, VMS (e.g. Gerritsen and Lordan, 2011)] and catches [electronic logbooks and fully documented fishing (Kindt-Larsen et al., 2011)] are becoming available. These new types of data have the potential to transform fisheries management. Rather than considering large management areas in annual cycles and on a single-stock basis, we may be able to use these technologies to move directly into a more spatiotemporally dynamic fisheries management approach [e.g. using real-time information for closures (Holmes et al., 2011; Needle and Catarino, 2011)], that integrates objectives for multiple commercial stocks with ecosystem considerations.

In this paper we describe a novel fishery management approach, which we call the ‘Real-Time Incentive (RTI)’ approach, in analogy to real-time closures (Needle and Catarino, 2011). In our approach, real-time spatial information is generally not used to close areas (although this possibility can be included), but instead incentives are provided to encourage fishing in certain areas rather than in others. The approach is implemented through quotas of ‘credits’, which we name ‘RTIs’, which are related to fishing opportunities through spatiotemporal tariffs. To illustrate how this approach might work we use historical data from the Irish and Celtic Seas and give two examples of different
management objectives that could potentially be addressed using this approach: i) managing cod mortality; ii) managing bycatch of protected elasmobranch species, while also managing cod mortality. Part and parcel of the approach is the risk estimation used for the setting as well as the results-based adjusting of the tariffs. We use the Irish and Celtic Seas’ cod fisheries for the illustration because in the case of the Irish Sea the current management plan for cod may actually allow Member States to unilaterally adopt our approach for (a segment of) their fleet [under the provision of Article 13 (Anonymous, 2008)], and in the case of the Celtic Sea our approach may be timely because a management plan for cod is currently under development with participation from the North Western Waters Regional Advisory Council (NWW RAC).

We discuss how the approach could be extended and adjusted to suit additional management requirements, such as mixed-fisheries and/or ecosystem considerations. Our proposed approach represents a radical departure from current management frameworks and accordingly we also discuss briefly some of the significant governance issues which may arise from it.

The Real-Time Incentives (RTI) approach

The basis of the approach is that the area would be divided up into ‘cells’ at a high spatial resolution. Each cell would have a certain ‘cost’ applied to fishing in that cell. These would be set by managers. Fishers would then ‘pay’ these costs in RTIs from their individual RTI account, allocated at the start of the management period, e.g. year. The costs, or tariffs (e.g. in RTIs per day), associated with fishing in each of the cells would be shown on colour-coded tariff maps. Using these maps, fishermen are then free to fish when and where they like as long as their RTI credit lasts and according to personal choice; they would not be allowed to exceed their annual RTI-quota once they have exhausted it. The total amount of RTIs annually available can be set in relation to (internationally agreed) objectives or targets of fishing mortality rate (or parts thereof if applied to fleet segments) of the stock of interest.

The cell tariffs could initially be set according to the historical spatial patterns of the catchability of the stock of interest. The tariffs can also be modified by expert biological knowledge (e.g. location of spawning grounds etc.) or based on stakeholder input. Additional spatial information about the ecosystem can be built into the tariffs depending on management objectives. For example, sites of interest such as cold water corals, marine mammal habitats, etc. could all be included. Cells with very high catchability of the targeted stock could have ‘infinite’ cost [i.e. they would be (temporarily) closed for fishing] for precautionary reasons. Cells could be closed for other reasons as well, e.g. if they were extremely vulnerable habitats. The rules used to weight the different sources of information to set the tariffs should be transparent and open to stakeholder participation in decision-making.

Tariffs could be updated on a given timescale. This temporal update could have different timescales for different factors – e.g. ‘real-time’ (say, weekly) update for the targeted
stock landings per unit effort (lpue) values, but annual update for habitat importance, or update when new information becomes available etc. For time-invariant factors there would be no update needed at all. The updating rules should also be transparent and open to participatory decision-making.

An essential part of the approach is that through simulations the tariffs can be related to levels of risk of under- and overshooting the various targets or objectives; these risk levels can be set explicitly by managers in a transparent way (reflecting societal choices). The approach is results-based: if it fails to deliver one or the other aim or objective on a particular time-scale, the tariffs can be adjusted up or down, e.g. at annual timescales.

Illustration of the RTI approach: Irish Sea and Celtic Sea

Managing the cod fishery

Following the method described by Gerritsen and Lordan (2011), each VMS record of Irish demersal otter trawlers (OTB) was allocated an effort value, which is the time since the previous VMS record (generally 2 hours). The VMS data were filtered for vessel speeds between 1.5 and 4.5 knots in order to select records corresponding to fishing activity. Gerritsen and Lordan (2011) have shown that vessel speed can distinguish fishing activity with an accuracy of 88%. Skippers of EC vessels of 10m in overall length are also required to record their retained catches on a daily basis (EEC, 1983). The daily retained catches (from here onwards referred to as ‘landings’) of cod were allocated equally to the ‘fishing’ VMS records for each vessel and date. The resulting cod landings and effort data were aggregated to a grid of 0.3° longitude * 0.2° latitude. Any grid cells with <20 h effort were omitted from the calculations involving the 2006-2009 data, but all information was retained for the weekly 2010 data (see below).

Using data from 2006-2009 we created ‘heat maps’ where the grid cells are coloured according to their lpue relative to the mean lpue of all grid cells with ≥20 h effort. The mean lpue was estimated separately for each of the two management areas, the Irish Sea and the Celtic Sea (Fig. 1a; for convenience, both are plotted on the same map). The relative lpue was divided into 6 arbitrary classes of: 0-0.1; 0.1-0.5; 0.5-1; 1-2; 2-5; >5 times the mean lpue. The chosen colours can be distinguished by red-green colour blind people. Because in this paper we only want to illustrate the approach rather than calculate values to be used in actual management, we pooled the data over the mesh-size groups. If this approach would be taken up as a cod-avoidance plan under the Article 13 provision of the Irish Sea cod management plan (Anonymous, 2008), calculations should be done separately for the mesh sizes of >100 mm (TR1) and 70-99 mm (TR2) of the OTB, and calculations could be done for the other regulated gear groups as well. Likewise, for the Celtic Sea separate calculations could be done by gear group and/or métier. These métiers or gear or mesh-size groups would then each have their own set of heat maps. In the current case of pooled OTB data, we pooled the data for the years 2006-2009 because the mean lpue did not differ between those years.
From these relative lpue maps baseline RTI-tariff maps are created, by translating the relative lpue classes into tariffs of 0.1; 0.5; 1; 2; and 5 RTI for the first 5 lpue classes and designating the grid cells with >5 times the mean lpue as closed areas; in addition, all grid cells for which no information exists get a tariff of 0.1 RTI (Fig. 1b). In this example, the tariffs are set ‘precautionary’ in that each tariff is the upper value of the range of the respective relative lpue, and the highest is set as closed to fishing. Risk analyses can be done (see below) so that managers can decide how to set the tariffs relative to the risk of over- or undershooting of the intended catch levels or fishing mortality rates they deem acceptable.

Figure 1. (a) 2006-2009 cod lpue relative to the mean, separately for the Irish Sea and the Celtic Sea; (b) RTI-tariffs; (c) RTI-tariffs for selective gear.

RTI-credits or RTIs can be seen as fishing-day equivalents in terms of mean fishing pressure. The total available amount of RTIs in a year follows from the Member State’s allocated fishing opportunity for cod. In the case of an agreed TAC and the Member State’s portion of it, a scientific advisory forecast could predict, based on historical data, how many average fishing days would be needed to take that catch; this could be done separately for the respective gear or mesh-size groups or métiers. In the case of a management plan with an effort regime the available RTIs might be directly related to the maximum allowable effort. In any case, each vessel gets assigned a number of RTIs equivalent to the number of average fishing days allowed. Throughout the year, fishers can then fish wherever they want, except in the closed areas, and they will have to pay 0.1 RTI, 0.5 RTI, 1 RTI, 2 RTIs, or 5 RTIs per day respectively from their account until their RTI allocation is exhausted. Thus, a fisher can fish in the white areas (which are areas for which either no information is available or the historic lpue is only up to 0.1 of the mean) and pay only 0.1 RTI per day. Expected cod landings will be low but the fisher can target other species and will not effectively be limited by effort restriction while fishing there. Alternatively, a fisher can fish in the red areas, where expected cod landings will be high, and because he will have to pay 5 RTIs there, his effort there will be heavily restricted by his available RTIs. In principle, controlling fishing activity by RTIs should be sufficient: no catch/landings quota should be necessary for the fleets/vessels operating under the scheme.
The approach allows for incentives for fishers to take up more selective gear by issuing different tariff maps for vessels using gear of which a scientific study has shown that it catches cod at much lower rates. For example, for a gear to which the cod vulnerability has been shown scientifically to be only 10% of that for the standard gear, the baseline tariff map (Fig. 1c) would allow fishing in almost all grid cells at tariffs of only 0.1 RTI or 0.5 RTI, which would effectively remove effort restriction. Ideally in this type of case, the burden of proof would be on the fisher to show that he was using the more selective gear. The default for tariff setting would be the least selective gear.

The basic approach described so far will only deliver in terms of controlling the fishing mortality to the levels intended by the TAC or other management measure if the historical spatial patterns of lpue are sufficiently predictive for the current year. The distribution of demersal species can vary with depth, bottom type, hydrological conditions, interactions with predators, prey and competitors, historical contingencies and other variables (Planque et al., 2011). Some of these are relatively constant over time, but others are not; and certainly in a migrating species spatial patterns cannot be expected to be constant over time. Therefore, in analogy to Real-Time Closures as applied in Scotland (Needle and Catarino, 2011), in our approach we need to update the tariffs with real-time information. Here we did this arbitrarily as follows. Imagine that the RTI system is in place for 2010, and that in the first week of fishing the tariffs are given by the baseline based on pooled data from 2006-2009 (Fig. 1b). The new information coming in from the first week of fishing in 2010 gives rise to a new lpue map (Fig. 2a). We developed an algorithm which looks up whether a given grid cell has information from the most recent fishing week. If so, if the relative lpue falls into the same class as on the old map, nothing happens. Conversely, if the relative lpue of the week belongs to a higher class, the new tariff will go up one class, and if the relative lpue of the week belongs to a lower class, the new tariff will go down one class. Letting tariffs go up or down by only one level at a time avoids that tariffs will be influenced too much by noise. Other algorithms can be envisaged, for example where a tariff goes up or down only if the relative lpue belonged to a higher or lower class for two consecutive weeks. In our imaginary example, in the second week of 2010 fishing is regulated according to the tariff map updated by information from week 1 (Fig. 2b). Then, the information coming in from the second week (Fig. 2c) is used by the same algorithm to update the map for the third week of fishing (Fig. 2d). Next, the information coming in from the third week (Fig. 2e) is used by the same algorithm to update the map for the fourth week of fishing (Fig. 2f). And so on and so forth. In this illustration the information we use is of course derived from a fishing season in the past (2010); however, in reality no fishing would take place in black cells so these could then not be updated by new information. A solution would be to reset the tariff of black cells to red after three weeks, analogous to the real-time closures in Scotland being closed for only three weeks (Needle and Catarino, 2011). Based on the available data for the first 18 weeks of 2010, the updated tariffs are depicted in Figure 3. Of course, this real-time approach will only deliver its intended results if the spatial patterns of week x are sufficiently predictive of those of week x+1.
Figure 2. (a, c, e) cod LPUE relative to the area-specific mean in weeks 1, 2, and 3 respectively; (b, d, f) RTI-tariffs for weeks 2, 3, and 4 respectively, updated by real-time information of weeks 1, 2, and 3 respectively.
Figure 3. RTI-tariffs for weeks 5 to 19, updated by real-time information from the week before, weeks 4 to 18 respectively.
Ecosystem considerations (elasmobranch conservation)

As an example of how the cod tariffs can be modified by ecosystem considerations we used the Irish data of observed trips spanning 1995-2011 (pooled). We considered the discards per unit effort (dpue) by grid cell of prohibited and no-catch elasmobranchs: Prohibited species: common skate (Dipturia batis), longnose skate (D. oxyrinchus), white skate (Rostroraja alba), and angel shark (Squalina squalina); No-catch species: spurdog (Squalus acanthias), porbeagle (Lamna nasus), and basking shark (Cetorhinus maximus) (Fig. 4a). Subsequently, we arbitrarily chose ‘black’ (closure) for all grid cells with >10 kg/h and ‘red’ (tariff of 5 RTI) for all grid cells with 1-10 kg/h and no colour for grid cells with <1 kg/h (Fig. 4b). Managers would make these choices based on their objectives and priorities. We then superimposed that map on top of the tariff map for cod such that a darker colour in the elasmobranch map overrides a lighter colour in the original tariff map (Fig. 4c). Again, managers may want to choose a different rule for

Figure 4. (a, b) dpue of prohibited and no-catch elasmobranchs (1995-2011), visually displayed as two different classifications of levels, with (b) reflecting the (fictive) management choice; (c) RTI-tariffs modified by elasmobranch discard information; (d) RTI-tariffs for cod-selective gear modified by elasmobranch discard information.
combining the two sources of information. In our example, the superimposition gives rise to additional closed and high-tariff grid cells because of high elasmobranch discard rates; we implicitly allow fishing in areas with low elasmobranch discard rates because those cells had no colour in the elasmobranch map (Fig. 4b) and therefore the original tariffs were not modified. Superimposing the elasmobranch map on top of the tariff map for selective gear, assuming that the properties that allow cod to escape do not affect the probabilities of elasmobranch capture, leads to a selective-gear tariff map with extensive white and yellow areas (with tariffs of 0.1 RTI and 0.5 RTI respectively) flanked by some red areas with tariffs of 5 RTIs and some black closed grid cells (Fig. 4d). It is very important to understand that the authors have chosen these calculation steps for illustration only. We are not proposing that elasmobranch discards are more or less important than cod catches. These choices should be made by stakeholders and managers and should reflect policy objectives which ultimately reflect choices of society.

Illustration of risk calculation for setting and adjusting tariffs

Now we illustrate the principles of how the risk of over- and under-catching the intended levels can be explored. Note that this simulation is not meant as a validation of the approach or of the choice of tariffs. Managers could use more sophisticated versions of these methods, when first setting the tariffs according to their objectives but also to adjust them in a results-based manner if they find that results were insufficient. We used the weekly tariffs as updated by the previous week’s information for the first 35 weeks of 2010 (as in Fig. 3; except that for the purposes of this illustration we treated the Celtic and Irish Seas as one). We simulated fishing in these 35 weeks, restricting activity to the area fished in 2006-2009. In this simulation (for simplicity) it is assumed that fishing trips are restricted to only one grid cell each in a given week. In reality these calculations can be done in a more sophisticated way to capture more uncertainty. We assigned trips randomly to the three-dimensional matrix formed by the available grid cells (those with data for 2006-2009; latitude and longitude representing two dimensions) and the 35 weeks (the third dimension), until 1000 RTIs were spent (this can be thought of as equivalent to 20 week-long average trips each by 50 boats) according to the weekly tariffs. The ‘actual’ relative landings were cumulated according to the actual relative lpue data for the respective weeks and cells. This procedure was repeated 1000 times. Figure 5 displays the frequency distribution of the actual landings relative to the intended landings (a value of 1 indicates that the actual landings were equal to the intended landings). It is clear that, in this case, the weekly spatial patterns predict the patterns of the following week quite well. In combination with setting the tariffs ‘precautionary’ (each tariff is given by the higher end of the range of relative lpue and the highest tariff represents closure), the risk of over-catching the intended landings level is approximately 25%, with a risk of undershooting the intended landings of around 75%. Nevertheless, in our example, landings are quite likely to be above 90% of the intended level and the risk of overshooting the intended level by more than 10% is very low (only ~3%). These probabilities are conditional upon the (unrealistic) assumptions within our illustration and should not be interpreted as true risks of the described approach. Managers could use similar simulations to find the tariff setting that suits their objectives best, even in cases
with less similarity in spatial pattern between consecutive weeks (in which case the spread of the histogram would be wider). Again it should be noted that choices of risk levels should be made by managers and should reflect policy objectives which ultimately reflect choices of society.

![Histogram of relative catch](image)

Figure 5. Frequency distribution of the (simulated) actual catch relative to the (simulated) intended catch.

**Discussion**

In this paper we have described a new approach to fisheries management that operates using readily available data and that works via a single indicator of fishing activity, the RTI fishing pressure unit.

The approach illustrated here hinges of course upon the availability of real-time information on effort and landings, or even catches. Although VMS data can be used as a proxy for spatial effort, these data are not without problems. For example, it cannot be unambiguously determined whether a vessel is fishing or not (Gerritsen and Lordan, 2011). Logbook information has its problems as well: it is not always reliable and not available on a real-time basis, but the biggest problem is that it generally refers only to retained catches or landings [since July 2011 some but not all discards have to be reported (Anonymous, 2009a)]. The development of electronic logbooks (e-logbooks) may solve some of these problems but it does not address the discard issue. Moreover, e-logbooks generally have a minimum temporal resolution of one day since records on a haul-by-haul basis are optional but not required, thus precluding fine-scale spatial
allocation of the (retained) catches. There may be some promise in the development of fully-documented fisheries by on-board cameras (Kindt-Larsen et al., 2011; Townsend et al., 2008), but this is very costly.

On the positive side, for our approach the VMS and e-logbook data would be available quickly and in digital form allowing a rapid update of the tariff maps. Enforcement and control of RTIs are much simpler than enforcement and control of catches or landings quota. Each fisher’s uptake of RTIs through time would be visible through VMS, by cumulating the numbers of hours at fishing speed weighted by the tariff of the grid cells. This would be visible to both the control agency and to the fisher who needs to keep track of his RTI-account. Exceeding the RTI-quota would be forbidden, and the control agency can confront fishers whose RTI-quota is used up when they are suspected to be fishing (based on VMS and e-logbooks).

If the tariffs are based on intended catch levels (including discards or under a discard ban) rather than on intended landings levels, the costs of monitoring the catches will be high. Perhaps a system can be envisaged [based on an idea by John Simmonds (see p. 35 in STECF, 2011)] where individual fleets or vessels can opt for high tariffs with low monitoring or lower tariffs with complete monitoring (fully documented or 100% observer coverage). For example, the respective tariffs of white, yellow, light-orange, dark-orange, and red would be 0.1 RTI, 0.5 RTI, 1 RTI, 2 RTIs, and 5 RTIs (as in the worked example) for the vessels that opt for complete but costly monitoring, and the respective tariffs would be 0.3 RTI, 1.5 RTIs, 3 RTIs, 6 RTIs, and 15 RTIs for vessels that opt for low-cost monitoring. These vessels would have 3 times less fishing opportunities but would not have as stringent and costly requirements to demonstrate their catches and/or show that they are not discarding (under a discard ban). Essentially, this would be a reversal of the burden of proof. Without any additional evidence we would assume they were fishing as before, but if the fishers could prove they were fishing more responsibly, they would then have access to a lower tariff.

If viewed as a single-species approach, the RTI-tariffs refer to real-time, high-resolution, relative cpue/lpue or catchability ('Fpue') of that single species. RTI-quotas are then equivalent to allowed average fishing-days in terms of fishing pressure. The total allowable RTIs can be calculated as the number of average fishing days needed to achieve an agreed (partial) F. Different tariff maps can be issued for different gear and mesh-size groups and métiers, as well as for vessels committing to the use of specially designed selective gear, e.g. for cod avoidance (in which case the burden of proof would be on the fisher). The tariffs might be modified by spatial discard information from observer trips of the single species concerned. Risk levels related to uncertainty can be set, according to policy decision, by adjusting the tariffs. We envisage the approach to be stand-alone for the fleets/vessels operating under the scheme, with no need for catch/landings quota. Nevertheless, actual removals must be monitored. If it turns out that the intended annual catch has been overshot by an extent that is of concern, for example because the catchabilities were higher than predicted, the tariff rates can be shifted upward accordingly for the next year. Within-year shifts would not be advisable as this
will result in unfair disadvantages (or advantages) for fishers who have based their business plan on delaying activity to later in the year.

For mixed fisheries and ecosystem considerations, the tariffs could integrate information on several commercial species, and, e.g., cetacean bycatch, habitat impacts, etc., as was illustrated with the example of elasmobranchs. In principle, the tariff maps can be modified and dynamically updated with any kind of information policy makers deem necessary. Fishery-dependent as well as fishery-independent information, and static, time-invariant, information as well as updatable information can be included, and the information can be weighted according to any objectives or aims set by policy. For example, certain information can be set to ‘override’ other information, as in the case of the black and red areas in our elasmobranch example. Alternatively, weights could be applied, such that cells can be adjusted to a slightly redder or yellower colouring depending on the information and the chosen weighting. It can be pointed out that the boundaries of cells where, e.g., elasmobranchs were discarded are too discrete. However, depending on the choice of acceptable risk level for the objective under consideration, neighbouring cells can get the same or similar tariffs and/or tariffs can be updated by annual or real-time information. In a results-based manner, the approach can be annually fine-tuned. Fitzpatrick et al. (2011) suggested a management approach which would combine VMS data with benthic impact models as a co-manageable solution to reversing the burden of proof in relation to fisheries benthic impacts. The approach proposed here is more inclusive in that in addition to addressing benthic impacts, it also deals with the other main environmental impacts of fishing, removal of commercial species and bycatch (OSPAR, 2010; Pascoe et al., 2009).

In a mixed-fishery approach that was explored by Rijnsdorp et al. (2007) at a temporal (seasonal) resolution of 12 months and a spatial resolution of four North Sea subareas, it was envisaged that the fishers would have multiple quotas of ‘fishing credits’, one quota each for every regulated species in the assemblage caught in the mixed fishery. The fishers would then be allowed to fish wherever and whenever they want so long as they would not exceed any of their credits-quotas (while obliged to land all fish of these species). Thus, a fisher could on a certain fishing trip spend 7 whiting credits and only 0.5 cod credit, while if he would make a different choice, i.e. to fish in a different subarea, he might spend only 3 whiting credits but 5 cod credits. The fisher would have to manage his multiple credits-quotas simultaneously by making his business decisions according to his own preferences. This system would reconcile the different fishing opportunities arising from different states of the stocks in the mixed assemblage – the age-old mixed-fisheries problem (Ulrich et al., 2011; Vinther et al., 2004) – by exploiting the knowledge of the spatiotemporal variation of the catchabilities of the species involved. Our proposed approach aims to capture similar ideas to Rijnsdorp et al. (2007) with a spatially and temporally explicit design, with “credits” for multiple targets, and a free choice for the fisher within that.

In the past decade several attempts have been launched trying to tackle the mixed-fishery dilemma, e.g. by models such as MTAC (Vinther et al., 2004), Fcube (Ulrich et al., 2011), and others (Rätz et al., 2007). Each of these proposed tools aims at a rational
compromise between the under-exploitation of some stocks and the over-exploitation of others. The compromise is calculated based on rules that reflect management choices, which are either set *ad hoc* (ICES, 2006; STECF, 2002; 2003; 2004; 2005; Ulrich et al., 2011; Vinther et al., 2004) or according to biological (ICES, 2006; Rätz et al., 2007; Ulrich et al., 2011) or economic objectives (Hoff et al., 2010; ICES, 2006; Ulrich et al., 2011). Such compromise approaches ignore the potential of adjusting the fishing activities based on the predictable spatiotemporal variation of the catchabilities of the species involved (as in Rijnsdorp et al., 2007), in a way that does not require a compromise between two undesirable outcomes. In principle, all quota can be taken exactly, but only if the fishers are incentivised to adjust their spatiotemporal fishing patterns to the spatiotemporal abundance-patterns of the various fish species. In a different approach (Gerritsen et al. submitted to IJMS), the spatial patterns in the species compositions of the landings are considered. These turn out to be relatively stable in time (at least in the specific case), opening avenues for spatial mixed-fisheries management.

The RTI system can be made completely transparent to the fishers. The fishers would have access not only to the weekly tariff maps, but also to the layers of considerations they are composed from. In our example, fishers would be able to see the map as based on the relative cod lpue and in addition they would see the map as based on elasmobranch discards, as well as the rule of combining them (in this case superposition with darker colours overriding lighter ones). This information can help the fisher decide on his tactics and make a fishing plan. This element of the approach also addresses the difficulty in linking individual vessel/skipper behaviour with wider ecosystem level impacts (Fitzpatrick et al., 2011).

If the approach were taken up, it would transform the practice of traditional stock assessment working groups. There will be a work load associated with the necessary calculations, not only of the tariffs based on multiple-species and ecosystem considerations but also of the tariff-related risks. The approach can be seen as a form of adaptive results-based management, because each year the tariffs can be adjusted and shifted as needed to “go in the right direction”.

The approach also addresses the current shift towards more decentralised, flexible and consensual styles of governance rather than top-down, centralised, command-and-control regulation. Stakeholder participation is thought to increase the sense of ‘ownership’ among fishers towards the overfishing problem and thereby the level of compliance to the regulations (Kraak, 2011). Giving flexible choices and responsibility to fishers may also help. Rather than the broad-brush, top-down prescriptions and restrictions of ‘one-size-for-all’ nature, the freedom and flexibility in the proposed system for individual fishers to choose and plan when, where, and how to fish, given the internationally agreed and biologically based constraints, makes it fit in very well with the EU aspirations. It corresponds well to the preference for “a fishery-by-fishery, incentive-driven, results-based approach as part of a more participative system of management” expressed by the industry via the North Sea Regional Advisory Council (NSRAC) in the recent evaluation of the EU cod plans (NSRAC opinion paper, p. 109 in STECF, 2011). Fishers and other stakeholders could also be involved in the determination of the algorithm for combining
layers to generate the tariff maps. For instance, in our example, they could propose that the two layers be averaged, rather than superimposed. If the targets for both cod and elasmobranchs were subsequently achieved this combination algorithm could be retained. If not, a more stringent combination could be tried. In this way, the RTI approach also goes a long way to providing a results-based management where each year’s outcome is used to modify the RTI algorithm for the following year. The approach therefore is inherently suited to co-management and could represent a pragmatic application of scientists and industry sharing the burden of proof in contrast to the probably unrealistic expectation that industry are in a position to unilaterally shoulder that responsibility.

In the RTI system the costs of fishing are internalized and have to be ‘paid’ (in terms of credits, equivalent to allowable fishing opportunities) by the fishers thereby creating the right incentives. Whereas in the current system the costs of discards, habitat destruction, or other ecosystem costs, are carried by society as a whole (including fishers and consumers in the future), in our proposed system these costs are automatically internalized to the fishing business: fishers have to take them into account in their business decisions. This way, fishers can be flexible in their options but directly responsible for the damage associated with certain activities; we can see no perverse incentives or opposing requirements as is frequently the case with current fisheries management regulations (STECF, 2011).

The RTI approach creates some obvious implications for fisheries governance. Many of these stem from our view that the approach is probably only implementable as a stand-alone management framework, at least for those fleet segments operating under the scheme, rather than an additional layer on top of existing input and output controls. The most glaring governance issue is whether the RTI approach would gain acceptance with the fishing industry. Our initial presentation to industry has been received with (cautious) interest. Industry representatives particularly liked that the approach allowed them freedom to fish as they wanted (within the constraints of the system). The other attractive aspect was that the system was simple in operation at the vessel level, in that they had only one measure (the RTIs) to consider, rather than the current mass of often conflicting measures. The potential for stakeholder input was also appreciated. More generally, they were also pleased that someone was thinking about alternatives to the current management approach, which they regard as largely discredited. Whether it is politically feasible to have the RTI approach applied by only one Member State’s fleet in a multinational fishery such as the Celtic Sea whitefish fishery while other fleets operate under traditional quota-based management is highly debatable and this may imply a required pan-European adoption of the approach. Nevertheless, as mentioned before, Member States might be able to adopt the approach for groups of vessels under Article 13 of the cod plan for the Irish Sea, North Sea, Kattegat, or West of Scotland stocks (Anonymous, 2008). Alternatively, should the RACs like the approach, a proposal could be made for suitable trial fisheries to test the approach. The impact of the approach on Relative Stability and related to that the tradeability of credits between fishermen and across fleets are also outstanding and topical issues. All of these considerations would require thorough investigation with industry and policy makers which are beyond the
scope of this paper. For now our intention is to present for discussion an approach to fisheries management which addresses many of the current challenges in implementing an ecosystem approach to fisheries management.

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