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Food for Thought

21st century fisheries management: a spatio-temporally explicit tariff-based approach combining multiple drivers and incentivising responsible fishing

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Traditionally fisheries management has focused on biomass and mortality, expressed annually and across large management units. However, because fish abundance varies at much smaller spatio-temporal scales, fishing mortality can potentially be controlled more effectively if managed at finer scale. The ecosystem approach requires more indicators at finer scales as well. Incorporating ecosystem targets would need additional management tools with potentially conflicting results. We present a simple, integrated, management approach that provides incentives for "good behaviour". Fishers would be given a number of fishing-impact credits, called real-time incentives (RTIs), to spend according to spatio-temporally varying tariffs per fishing day. RTI quotas and tariffs could be based on commercial stocks and ecosystem targets. Fishers could choose how to spend their RTIs, e.g. by limited fishing in high-catch or sensitive areas or by fishing longer in lower-catch or less sensitive areas. The RTI system does not prescribe and forbid, but instead allows fishers to fish wherever and whenever they want; ecosystem costs are internalized and fishers have to take them into account in their business decisions. We envisage no need for traditional landings or catch quotas for the fleets while operating under the scheme. The approach could facilitate further devolution of responsibility to industry.

Keywords: discards, ecosystem approach to fisheries management (EAFM), fisheries management, incentives, internalised costs, real-time information, spatio-temporal flexibility, tariffs.

Introduction

In this study, 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, an adaptive and results-based management approach, and a more devolved and less top–down management.

We do not propose this approach as a panacea to solve all problems in all fisheries. It is presented as a mean to accommodate the complexities inherent in combining single and multispecies fisheries management with an ecosystem approach. How applicable it will be will depend on the ecological, economic, and social characteristics of a particular fishery. Additionally, and in keeping with the fact that the paper is in the Food for Thought section, we are not prescriptive about the specific application of our approach as this would also be case-dependent and would require a detailed assessment by relevant stakeholders and policy-makers on the necessary value judgements and preferred trade-offs for a particular fishery. Similarly, a detailed management strategy evaluation (MSE) type assessment of a particular fishery is beyond the scope of this paper, but given that the approach has already attracted the interest from members of the North Western Waters Regional Advisory Council (NWWRAC) and from the Scientific, Technical, and Economic Committee for Fisheries (STECF), we intend to follow-up with such an evaluation.

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 single-species landings quotas (total allowable catches, TACs) along with effort and capacity controls and technical measures. But these tools are inadequate for an ecosystem approach to fisheries management (EAFM), where we would have to include mixed-fisheries and 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 (Garcia and Cochrane, 2005; Crowder *et al*., 2008). In parallel with these ongoing challenges in moving towards EAFM, a paradigm shift is occurring from top–down regulation (Daw and Gray, 2005) towards strengthened stakeholder participation in fisheries governance, reflected in the Green Paper of the European Commission (EC) on the new Common Fisheries Policy (CFP; Anon., 2009b) and the EC's proposal for the new CFP (Anon., 2011). The EC Green Paper essentially proposes devolution towards more regional-based management and to include all 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 utilized for fisheries assessment and management. For example, real-time high-resolution spatio-temporal data on vessel activity [vessel monitoring systems, VMSs (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 spatio-temporally 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. It would enable the directing of fishing effort at a fine spatio-temporal scale such that species with ample agreed fishing opportunities can be targeted while avoiding concentrations of species with low fishing opportunities and/or otherwise vulnerable areas.

In this paper, we describe our approach, which we call the "real-time incentive (RTI)" approach, in analogy to real-time closures (Needle and Catarino, 2011). In our approach, however, 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 fishing-impact credits, which we name "RTIs", which are related to fishing opportunities of commercial stocks and ecosystem targets through spatio-temporal tariffs. The approach does not "provide" (rationales for) the (ecosystem) objectives, targets, and fishing opportunities; these need to be decided upon through the appropriate governance scheme, e.g. the EU. Similar approaches have been proposed and explored, such as the individual habitat quota system for habitat conservation (Holland and Schnier, 2006a, b) and the individual transferable effort system implemented for the Australian Eastern tuna and billfish fishery (Pascoe *et al*., 2010) incentivising lower bycatch of vulnerable species through spatially varying multipliers of allowed effort. Our approach goes beyond these, because it proposes an explicit implementation for the regulation of the actual fishing opportunities on commercial (target) stocks, into which the ecosystem considerations can be integrated through the use of a single credit currency.

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 be addressed using this approach: (i) managing cod (*Gadus morhua*) 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 for 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 (Anon., 2008)], and for the Celtic Sea, our approach may be timely because a management plan for cod is currently under development with participation from the NWWRAC.

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 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 costs 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, fishers are then free to fish when and where they choose as long as their RTI credit lasts; they would not be allowed to exceed their 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 the fishing mortality rate (or parts thereof if applied to fleet segments) of the stock of interest. There would be no catch or landings quota of the stock(s) of interest for fisheries while operating under the RTI scheme.

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.) and/or based on stakeholder input. Additional spatial information about the ecosystem can be built into the tariffs depending on management objectives. For example, vulnerable areas, such as cold water corals, nursery areas, spawning grounds, marine mammal hotspots, could all be included. The approach also allows for the (temporary) closure of particular cells or groups of cells. For example, this could be where there had been a recent, particularly high catch of a species of interest, similar to the real-time closures used for cod in the North Sea (Holmes *et al.*, 2011). Closures could also be applied to protect vulnerable ecosystem components (e.g. cold water corals) or for any other agreed rationale. Such closures can effectively be seen as applying an "infinite" tariff. As in all aspects of the approach, such decisions should be transparent and involve all stakeholders, not just fishers.

Tariffs could be updated on any chosen time-scale. This temporal update could have different time-scales for different factors—e.g. "real-time" (say, weekly) update for the stock of interest by landings per unit effort (lpue) values, but possibly annual updates 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, involving expert opinion and stakeholders in a clear and unbiased process.

An essential part of the approach is that through simulations the tariffs can be related to the 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 adaptive: 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, for example, at annual time-scales.

Illustration of the RTI approach: Irish Sea and Celtic Sea

Managing the cod fishery

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, as described below, separately for each of the two stocks, the Irish Sea cod and the Celtic Sea cod (Figure 1a; for convenience, both are plotted on the same map). The relative lpue was divided into six arbitrary classes of: 0–0.1, 0.1–0.5, 0.5–1, 1–2, 2–5, and >5 times the mean lpue.

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 h). The VMS data were filtered for vessel speeds between 1.5 and 4.5 knots to select records corresponding to fishing activity. Gerritsen and Lordan (2011) have shown that vessel speed can distinguish fishing activity of demersal otter trawlers with an accuracy of 88%. Skippers of EC vessels of ≥10 m in overall length are also required to record their retained catches daily (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).

Because in this study 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 were taken up as a cod avoidance plan under the Article 13 provision of the Irish Sea cod management plan (Anon., 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 (a métier is defined as: "a group of fishing operations targeting a similar (assemblage of) species, using similar gear, during the same period of the year and/or within the same area and which are characterized by a similar exploitation pattern"; ICES, 2003). 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 over the years 2006–2009.

From these relative lpue maps, baseline RTI-tariff maps were created, by translating the relative lpue classes into tariffs of 0.1, 0.5, 1, 2, and 5 RTIs for the first five lpue classes and designating the grid cells with >5 times the mean lpue as closed areas; in addition, all grid cells for which no cod lpue information exists got a tariff of 0.1 RTI (Figure 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 tariff levels relative to the risk of over- or undershooting of the intended catch levels or fishing mortality rates they deem acceptable.

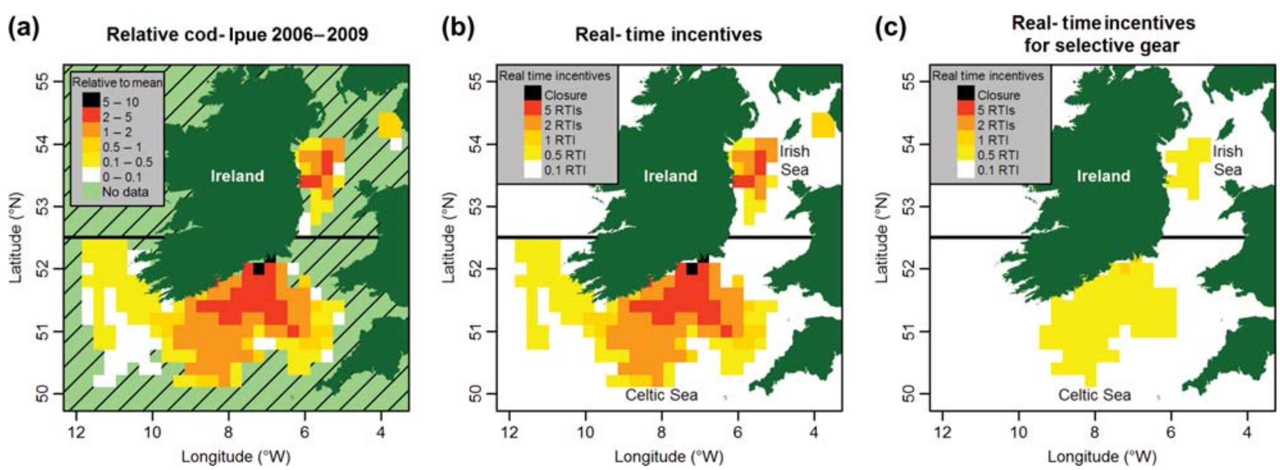

Figure 1. (a) 2006–2009 cod lpue relative to the mean, separately for the Irish Sea and the Celtic Sea; (b) RTI tariffs; (c) RTI tariffs for selective gear.

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. For 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. For 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, 0.5, 1, 2, or 5 RTIs per day, respectively, from their account until their RTI allocation is exhausted. Therefore, a fisher can fish in the white areas (which are areas for which either no cod catch information is available or the historical 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 while operating under the scheme.

Incentives for the use of selective gear

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 (Figure 1c) would allow fishing in almost all grid cells at tariffs of only 0.1 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.

Real-time tariffs

The basic approach described so far will only deliver in terms of controlling the fishing mortality to the levels intended by the TACs 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 (Petitgas *et al.*, 2010), and other variables (Planque *et al.*, 2011). Some of these factors 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 to 2009 (Figure 1b). The new information coming in from the first week of fishing in 2010 gives rise to a new lpue map (Figure 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 the issue 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 (Figure 2b). Then, the information coming in from the second week (Figure 2c) is used by the same algorithm to update the map for the third week of fishing (Figure 2d). Next, the information coming in from the third week (Figure 2e) is used by the same algorithm to update the map for the fourth week of fishing (Figure 2f), etc. 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 not be updated by new information. A solution would be to reset the tariff of black cells to red after 3 weeks, analogous to the real-time closures in Scotland being closed for only 3 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 $x + 1$.

Ecosystem considerations (elasmobranch conservation)

As an example of how the cod tariffs can be modified to incorporate other ecosystem elements, we used the Irish data of observed OTB trips spanning 1995–2011 (pooled). We considered the discards per unit effort (dpue) by grid cell of a number of species of vulnerable elasmobranchs: common skate (*Dipturus batis*), long-nose skate (*D. oxyrinchus*), white skate (*Rostroraja alba*), angel shark (*Squalina squalina*), spurdog (*S. acanthias*), porbeagle (*Lamna nasus*), and basking shark (*Cetorhinus maximus*; Figure 4a). Subsequently, as an intermediate step, we created a rule whereby we arbitrarily chose "black" (closure) for all grid cells with >10 kg h^{-1}, "red" (tariff of 5 RTIs) for all grid cells with 1–10 kg h^{-1}, and no colour for grid cells with <1 kg h^{-1} (Figure 4b). Managers would make these choices based on their conservation objectives and priorities. Finally, 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 (Figure 4c). Again, managers may want to choose a different rule for combining the two sources of information (e.g. a weighted sum or a weighted average). 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 received no colour in the intermediate-step elasmobranch map (Figure 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

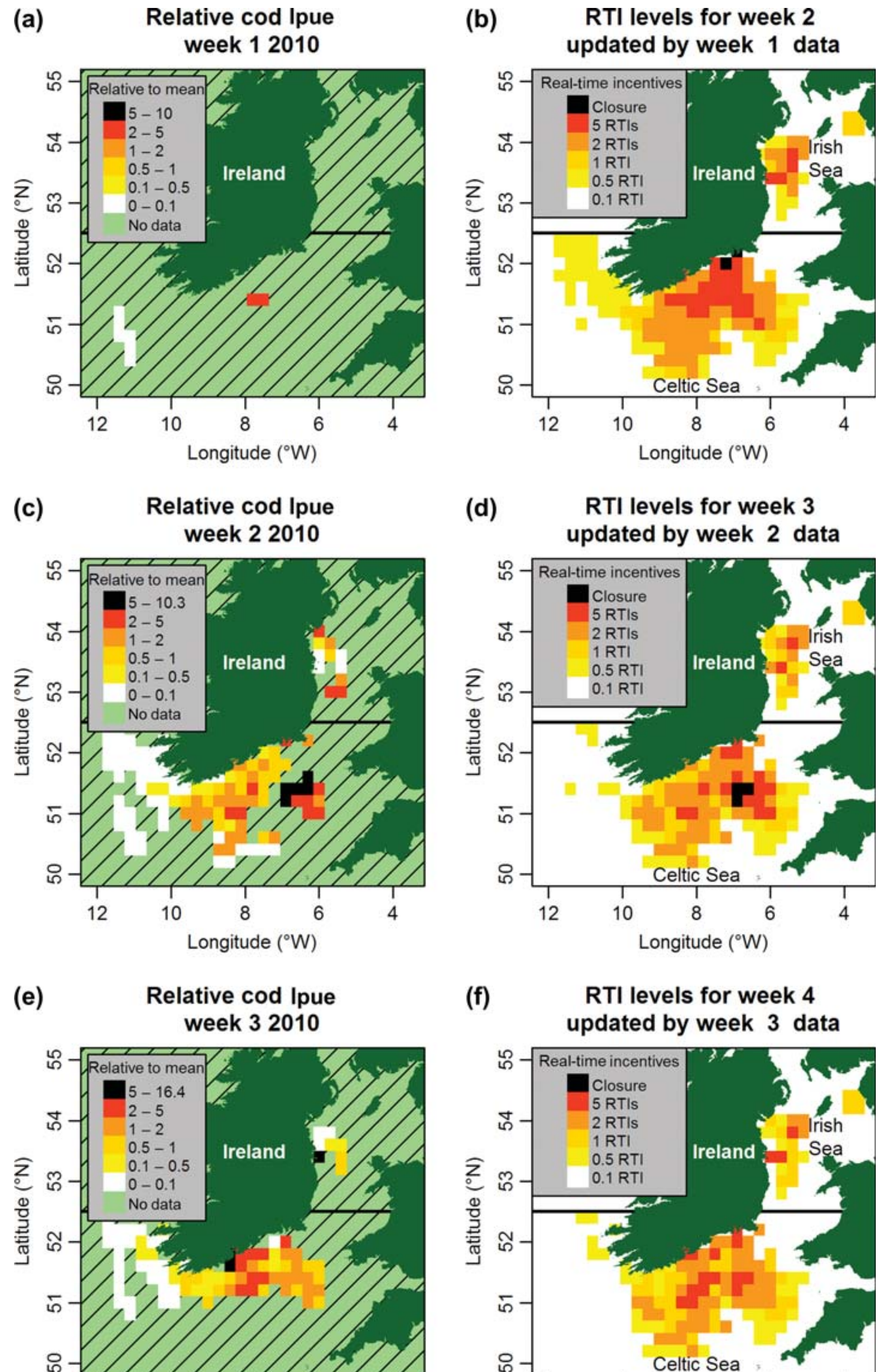

Figure 2. (a, c, and e) Cod lpue relative to the area-specific mean in weeks 1, 2, and 3, respectively; (b, d, and f) RTI tariffs for weeks 2, 3, and 4, respectively, updated by real-time information of weeks 1, 2, and 3, respectively.

yellow areas (with tariffs of 0.1 and 0.5 RTI, respectively) flanked by some red areas with tariffs of 5 RTIs and some black closed grid cells (Figure 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 societal choices.

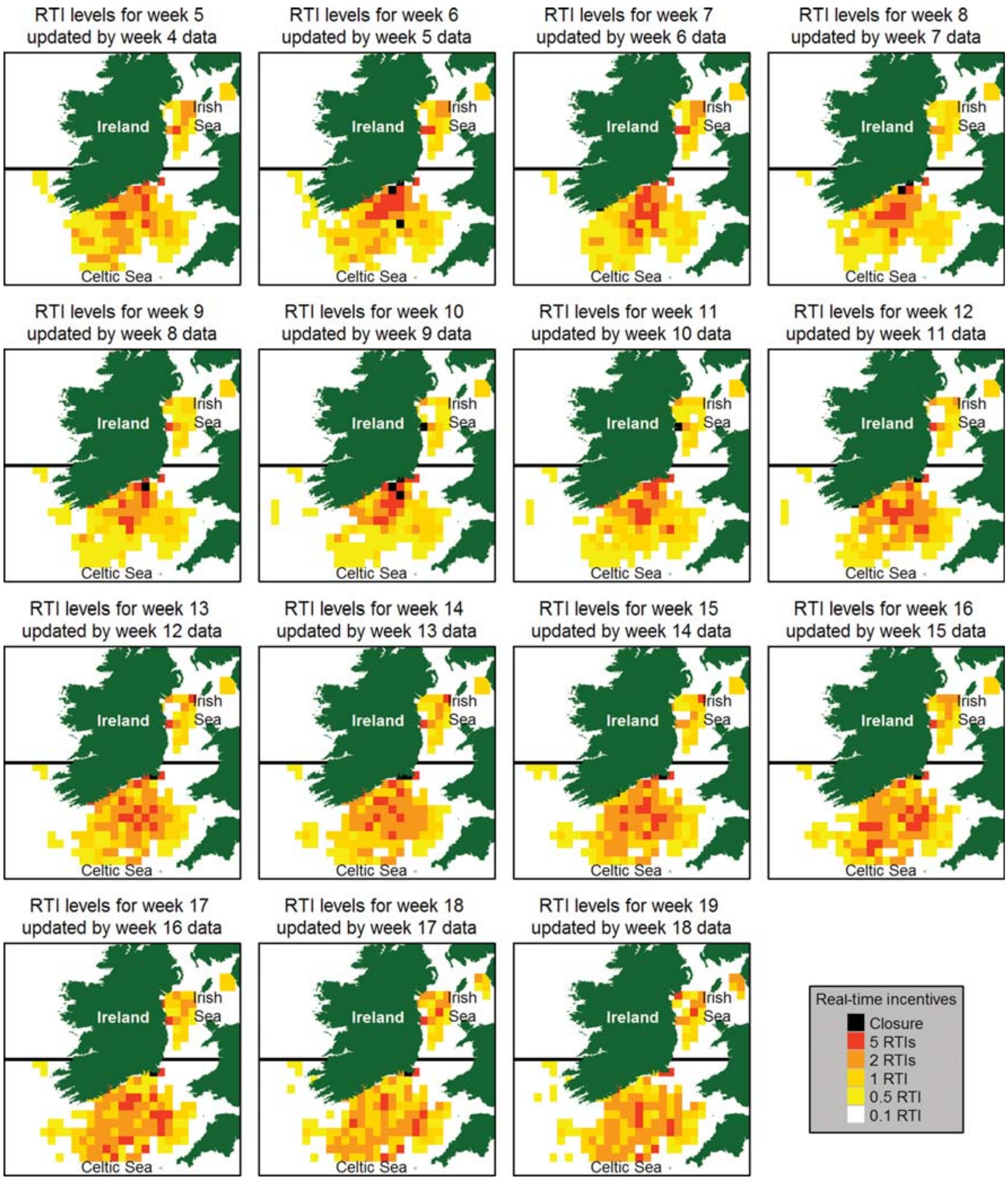

Figure 3. RTI tariffs for weeks 5–19, updated by real-time information from the week before, weeks 4–18, respectively.

Illustration of risk calculation for setting and adjusting tariffs in relation to target species

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 adaptively if they find that results did not achieve the objectives. We used the weekly tariffs, as updated by the previous week's information, for the first 35 weeks of 2010 (as in Figure 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, it is assumed (for simplicity) 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" landings were accumulated according to the actual relative lpue data for the respective weeks and

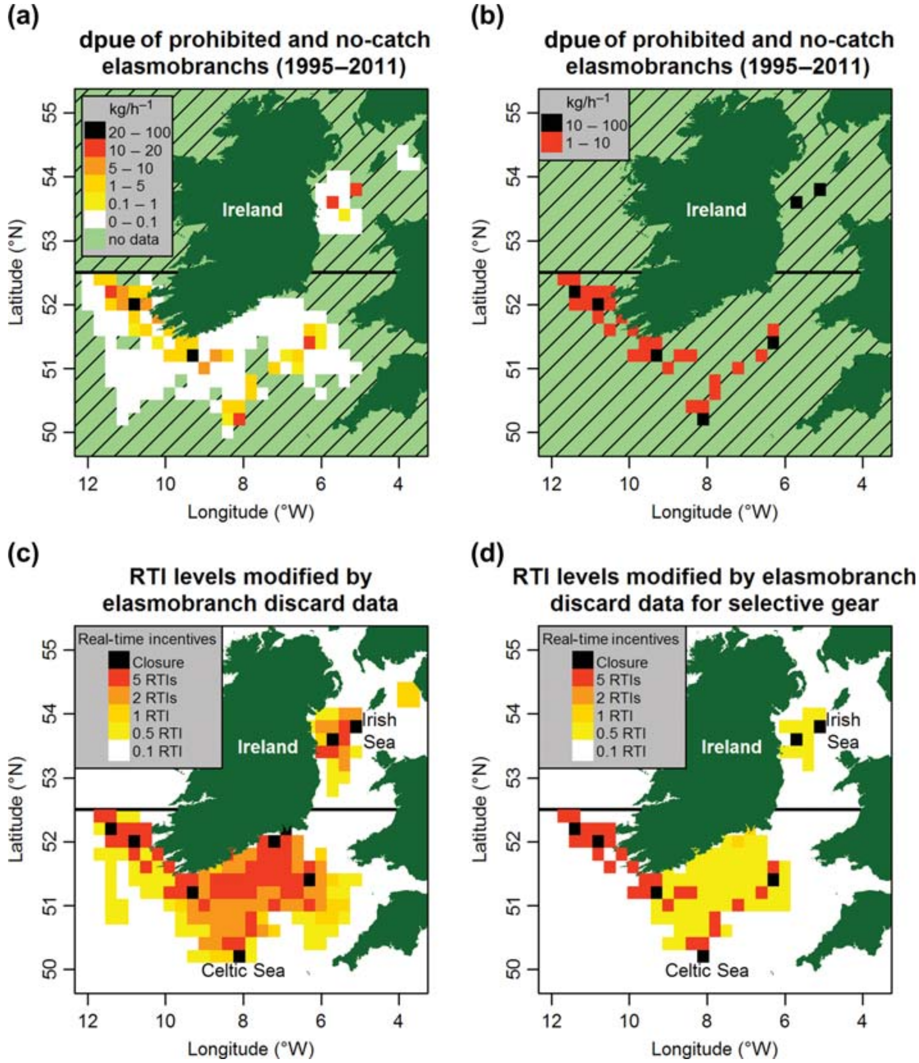

Figure 4. (a and b) dpue of vulnerable elasmobranchs (1995–2011), visually displayed as two different classifications of levels, with (b) reflecting the (fictitious) management choice; (c) RTI tariffs modified by elasmobranch discard information; (d) RTI tariffs for cod-selective gear modified by elasmobranch discard information.

cells, and the "intended" landings were set to 1000. 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 "precautionary" tariffs (each tariff is given by the higher end of the range of relative lpue except for the highest tariff, which represents closure), the risk of overcatching the intended landings level is ~25%, with a risk of undershooting the intended landings of around 75%. Nevertheless, in our example, landings are 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 (Kraak *et al.*, 2010). Managers could use similar simulations to find the tariff setting that suits their objectives best, even in cases with less similarity in the spatial pattern between consecutive weeks (in which case the spread of the histogram would be wider). Before actual implementation, an initial impact assessment is recommended, with full MSE simulations with different plausible scenarios, for example, of fisher behaviour. 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.

The approach can be envisaged as adaptive: 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 (e.g. resulting from increased efficiency), the tariff-level can be shifted upwards accordingly for the next year. Within-year shifts of the mean tariff level would not be advisable (except for emergencies) 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.

A similar approach could be incorporated into examining the rate of elasmobranch catch and discarding. If the levels were deemed to have changed and, in particular, increased from what was previously known, the tariffs could be increased to reflect elasmobranch vulnerability.

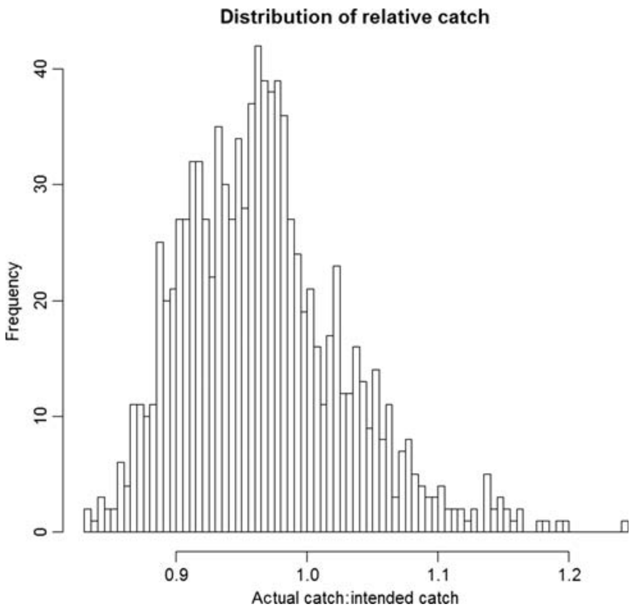

Figure 5. Frequency distribution of the (simulated) actual catch relative to the (simulated) intended catch.

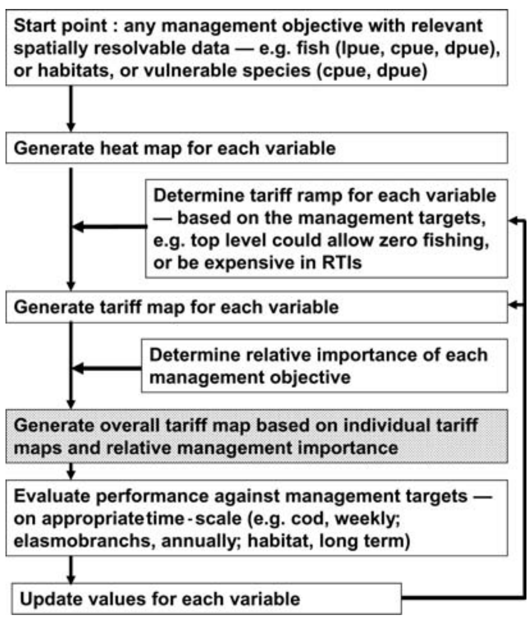

Figure 6. Flow chart giving an overview of the operation of the RTI approach. The stippled box represents the management measure issued to fishers, namely the tariff map.

Figure 6 displays a flow chart giving an overview of the operation of the RTI approach.

Discussion

Fisheries management currently operates in a domain of unrealistic precision (Hauge, 2011) based on the micro-management of some aspects (the exploitation level of commercial target stocks based on landings quotas), while effectively ignoring others (unintended mortality of vulnerable species, collateral damage to vulnerable habitats, discarding, and incompatibility of exploitation patterns in mixed-species fisheries). We consider this paradigm to be incompatible with effective ecosystem-based fisheries management. In this study, 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-impact unit. We propose that it should be possible to manage the pressure and impact of fishing on commercial stocks and on the ecosystem, by managing the cost and allocation of RTIs. We do not propose a panacea that can be applied across all fisheries in the world; we believe that the approach is appropriate to fisheries where sufficiently temporally and spatially resolved catch data are available.

Real-time data for the setting of tariffs and the keeping track of RTI uptake

The approach illustrated here hinges upon the availability of real-time spatial information on effort and landings or, ideally, catches. Although VMS data can be used as a proxy for spatial effort (e.g. for trawlers), 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 currently not available on a real-time basis, but the biggest problem is that it generally refers only to retained catches or landings [however, since July 2011, some but not all discards have to be reported (Anon., 2009a) and the requirement to report or even ban discards will likely increase in the future (Anon., 2011)]. The development of electronic logbooks (e-logbooks) may solve some of these problems but it does not address the discard issue. There may be some promise in the development of fully documented fisheries by on-board cameras (Townsend *et al*., 2008; Kindt-Larsen *et al*., 2011).

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 single RTIs are logistically simpler than enforcement and control of multiple catches or landings quotas. Each fisher's uptake of RTIs through time would be visible through electronic reporting of spatial effort. This visibility would be 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. We recognize that changing the enforcement from "evidence of landings" to "observation of fishing" could require some change in the laws used to enforce the control measures, this would require scrutiny from a legal perspective, and the implication (if any) may vary between countries.

Monitoring, tariffs, incentives, and reversal of the burden of proof

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, 0.5, 1, 2, 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, 1.5, 3, 6, and 15 RTIs for vessels that opt for low-cost monitoring. These vessels would have three times fewer fishing opportunities but would not have as stringent and costly requirements to demonstrate their catches and/or show that they are not discarding (in the case of a discard ban). Essentially, this would be a reversal of the burden of proof. Without any additional evidence, it would be assumed that 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 and thus effectively have increased allowed fishing effort available. Although it may seem difficult to set the tariffs relative to each other initially (e.g. based on the relative uncertainties), they can be adaptively adjusted based on past results, e.g. annually. Responsible fishing could also be incentivised and incorporated into "green" labelling schemes, such as that run by the Marine Stewardship Council.

Multiple drivers

For a single species, the RTI tariffs refer to real-time, high-resolution, relative cpue/lpue or catchability ("Fpue") of that species and RTI quotas are equivalent to allowed average fishing-days in terms of allowed fishing pressure for that species; if based on lpue data, the tariffs might be modified by spatial discard information from observer trips. For mixed fisheries and ecosystem considerations, the tariffs could integrate information on several commercial species, and, for example, cetacean and seabird bycatch, habitat impacts, high discard rates, as was illustrated with the example of elasmobranchs.

Ecosystem impacts

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 "overrule" other information, as for 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, for example, elasmobranchs were discarded are too discrete. However, depending on the choice of the 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 an adaptive manner. It is important that multiple spatial and temporal scales are included for a wide range of non-target species and habitat types. For example, some seabird and pelagic dolphin species range widely, whereas others occur in genetically distinct populations, often on a regional seas basis (Mirimin *et al.*, 2009, 2011). While identifying distributional "hotspots" on a seasonal basis is feasible, total anthropogenic removals would need to be assessed on a wider scale, along with the discrete units described here. The scales used for both the target species and non-target species must be explicit to ensure that management units overlie relevant ecological processes (Crowder *et al.*, 2008).

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 (Pascoe *et al.*, 2009; OSPAR, 2010).

Mixed fisheries

In a mixed-fisheries 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 credit quotas (while obliged to land all fish of these species). Therefore, a fisher could, on a certain fishing trip, spend, for example, 7 whiting (*Merlangius merlangus*) credits and only 0.5 cod credit, whereas if he were to 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 credit 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 (Vinther *et al.*, 2004; Ulrich *et al.*, 2011)—by exploiting the knowledge of the spatio-temporal variation in the catchabilities of the species involved.

In the past decade, several attempts have been made to tackle the mixed-fishery dilemma, for example, 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 underexploitation of some stocks and the overexploitation of others. The compromise is calculated based on rules that reflect management choices, which are either set *ad hoc* (STECF, 2002, 2003, 2004, 2005; Vinther *et al.*, 2004; ICES, 2006; Ulrich *et al.*, 2011) or according to biological (ICES, 2006; Rätz *et al.*, 2007; Ulrich *et al.*, 2011) or economic (ICES, 2006; Hoff *et al.*, 2010; Ulrich *et al.*, 2011) objectives. Such compromise approaches ignore the potential of adjusting the fishing activities based on the predictable spatio-temporal variation in the catchabilities of the species involved (as in Rijnsdorp *et al.*, 2007), in a way that does not require a compromise between undesirable outcomes. Note that fishers are able to fine-tune their targeting behaviour and avoid over- and underutilization of quotas if the incentives to do so are there (Branch and Hilborn, 2008). In current management, where overquota discarding is legal, no such incentive exists because costs of discarding are not internal to the fishing business. In principle, all quotas can be taken exactly, but only if the fishers are incentivised to adjust their spatio-temporal fishing patterns to the spatio-temporal abundance patterns of the various fish species.

No single generic recommendation can be made at present for how to manage the mixed fisheries: by multiple species-specific RTI quotas (analogous to Rijnsdorp *et al.*, 2007) or by combining fishing opportunities for several species into one RTI system; this choice should be made by the policy-makers in consultation with stakeholders. Multiple RTI quotas would ensure that each species or impact is more tightly controlled without having to find some weighting scheme to create a single RTI that will balance impacts on different species and ecosystem components and still allow some fishing to take place. On the other hand, the system with multiple RTI quotas would become overly complicated for the fishers to deal with, whereas the advantage of the system with only one currency is its simplicity. In the latter case, control of each impact would be less precise, and a compromise between conflicting concerns would have to be accepted; managers would have to decide on the appropriate weighting about the risk of under- and overutilization of the respective fishing opportunities. Again, these decisions should be made by the policy-makers in consultation with stakeholders. An advantage of the use of a single currency may actually be that it imposes the demand to explicitly trade-off or prioritize different impacts. The problem when conflicting or non-prioritized impacts are not explicitly handled by the (management) system is that it may result in a highly complex implementation, which is not transparent in its objectives. In either case (multiple or single RTI quota), contradicting incentives are avoided: in the first case, the fishers are incentivised to target optimally, and in the second case, there is only one quota to adhere to, while individual flexibility is maintained. Alternatively, managers could consider the most restrictive objective(s) (the species for which the least fishing effort is allowed), e.g. cod, and set the (multiple) RTI quota accordingly, while managing the other species by the traditional quotas. Management area-specific solutions must be worked out, preferentially in a participatory governance manner.

Transparency, fisher buy-in, stakeholder participation, and governance

The RTI system can be made completely transparent to the fishers and other stakeholders. The fishers and other stakeholders would have access not only to the weekly tariff maps, but also to the layers of considerations they are composed from. In our example, they 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).

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 discarding, 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.

The approach also addresses the current shift towards more decentralized, flexible, and consensual styles of governance rather than top–down, centralized, command-and-control regulation. 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, within the (internationally) agreed and biologically based constraints, makes it fit in very well with the EU aspirations (Anon., 2009b). 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, then a more stringent combination could be tried. In this way, the RTI approach also goes a long way to providing adaptive 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 managers, scientists, and stakeholders sharing the burden of proof in contrast to the probably unrealistic expectation that industry is in a position to unilaterally shoulder that responsibility.

Our initial presentation to the fishing industry (Irish Fishery Science Partnership, Dublin, 8 July 2011) has been received with interest. Industry representatives particularly liked that the approach allowed them flexibility in fishing tactics (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 single RTIs) to consider, rather than what they see as the current mass of often conflicting measures (see also opinion papers of the RACs in STECF, 2011). 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.

In relation to strategic behaviour by fishers in tariff-setting and trade-off negotiation, the approach faces the same problems as any other but has the advantage of explicit discussion between all relevant participants from the outset of the required targets, desired objectives, and underlying criteria used in deciding on weightings and specific application of the RTI system. While not claiming that the system is impervious to strategic actions by stakeholders, it does contrast greatly with the current approach in many European fisheries which suffer from *inter alia* a lack of clarity in objectives and incentives for responsible stakeholder actions (Anon., 2009b). Another positive factor is that non-permanent spatial approaches to fisheries management, such as the Celtic Sea cod and herring spawning box closures, continue to be well supported by the fishing industry.

The RTI approach creates some obvious implications for fisheries governance. Many of these stem from our view that the approach can be implemented as a stand-alone management framework, at least for those fleet segments operating under the scheme, rather than an additional layer on top of the existing input and output controls. 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. Of course solutions can be envisaged; for example, where a Member State allows fleet segments to operate under the RTI scheme for part of the year but still maintains the overall national quota as the annual limit. 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 (Anon., 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 tradability of credits between fishers and across fleets are also outstanding and topical issues. We leave it open whether RTIs should be tradable or not; both options are possible. The discussions on the pros and cons of tradability of ITQs apply similar to RTIs. All these considerations would require thorough investigation with industry and policy-makers which are beyond the scope of this study. For now, our intention is to present for discussion an approach to fisheries management which addresses many of the current challenges in implementing an EAFM.

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