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Unsuitability of TAC management within an ecosystem approach to fisheries: An ecological perspective

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ABSTRACT

Fisheries management in European waters is gradually moving from a single-species perspective towards a more holistic ecosystem approach to management (EAM), acknowledging the need to take all ecosystem components into account. Prerequisite within an EAM is the need for management processes that directly influence the ecological effects of fishing, such as the mortality of target and non-target species. Up until recently, placing limits on the quantities of fish that can be landed, through the imposition of annual total allowable catches (TACs) for the target species, has been the principal management mechanism employed. However, pressure on non-target components of marine ecosystems is more closely linked to prevailing levels of fishing activity, so only if TACs are closely related to subsequent fishing effort will TAC management serve to control the broader ecosystem impacts of fishing. We show that in the mixed fisheries that characterise the North Sea, the linkage between variation in TAC and the resulting fishing effort is in fact generally weak. Reliance solely on TACs to regulate fishing activity is therefore unlikely to mitigate the impacts of fishing on non-target species. Consequently, we conclude that the relationship between TACs and effort is insufficient for TACs to be used as the principal management tool within an EAM. The implications, and some alternatives, for fisheries management are discussed.

1. Introduction

Fishing is one of the most widespread anthropogenic activities within the marine environment, and one that has perhaps the greatest impact on marine ecosystems. Direct and indirect effects of fishing have resulted in overfishing of target fish stocks, increased mortality of non-target by-catch species, destruction of benthic habitats, and consequent changes in the structure and functioning of faunal communities (e.g. Jennings and Kaiser, 1998; Daan et al., 2005). Traditional fisheries management aims to regulate the exploitation of a few commercially important species, but generally does not consider the overall impact of fishing on other ecosystem components. However, over the last 15 years the emphasis of fisheries management has gradually changed from management of individual target species towards a more holistic approach with the maintenance of the healthy functioning of the entire ecosystem as a primary goal (FAO, 2003). The UN Conference on the Environment and Development (1992) emphasized in Agenda 21 that the protection of the marine ecosystem and the use of marine resources were inseparable and that new approaches were needed. Consequently, the concept of an Ecosystem Approach to Management (EAM) was promoted and it is now being incorporated into European marine management strategies to meet objectives set for 2010 (e.g. ICES, 2005). Numerous publications have outlined and defined the overall objectives and commitments for the EAM (reviewed in e.g. Browman and Stergiou, 2004b; Browman and Stergiou, 2005), but the future success of the EAM will strongly depend on whether these rather generic and conceptual commitments can be channelled into specific management guidelines (Jennings, 2004; Garcia and Cochrane, 2005; Gislason, 2006).

It is clear that indicators have a key role to play in the implementation of an EAM (e.g. Jennings, 2005; Rees et al., 2008). Various types of indicator will be required: indicators of the pressure imposed by anthropogenic activities on components of the marine ecosystem

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(Piet et al., 2007); indicators that quantitatively describe changes in the state of various aspects of individual ecosystem components affected by anthropogenic activity (e.g. Greenstreet and Rogers, 2006); and indicators that quantify the management response, intended to modify the pressure on the system imposed by the activity in question so as to achieve particular objectives for the state of different ecosystem components. In order for an indicator based EAM to be successfully implemented, these various types of indicator will have to applied as sets within frameworks, such as the Pressure-State-Response (PSR) framework (Garcia and Staples, 2000), which explicitly link the coupled indicators through well understood relationships that can support predictive quantitative modelling. The current fisheries management process in fact provides an excellent example of the PSR framework in action. Here spawning stock biomass (SSB) takes on the role of "state" indicator and fishing mortality (f) acts as the "pressure" indicator, with both indicators linked through the theoretical virtual population analysis (VPA) model. Fisheries managers then attempt to attain the desired level of SSB ("state" of the stock) through direct manipulation of landings, so that total allowable catch (TAC) effectively becomes the "response" indicator. From an ecosystem perspective, the EAM should aim to reduce the impact of fishing not only on target stocks, but also on non-target species (fish and invertebrate benthos), habitat structure and ecological interactions and functioning. Thus, the question arises whether current management strategies are using appropriate (pressure, state and response) indicators for an implementation of the EAM.

Within the fisheries management context, data are routinely collected to estimate f each year, and so allow the effect of fishing "pressure" on the "state" of commercial fish stocks to be modelled using VPA or similar population dynamics models. For most other components of the marine ecosystem, however, the data necessary to determine the mortality caused by fishing are rarely, and may never, be available; by-catch rates of cetaceans, seabirds, or benthic invertebrates are not routinely reported or recorded. Estimating the "pressure" from fishing on these wider aspects of marine ecosystems will therefore rely on modelling anthropogenic induced mortality, with the models applied to measures of "activity", such as fishing effort. In essence the PSR framework needs to be extended to an Activity-Pressure-State-Response (APSR) framework with explicit models developed that quantitatively describe the relationship between activity levels and resulting pressure (Piet et al., 2007; Greenstreet et al., 2009). This therefore raises an important question, is management through catch limitation by TACs the appropriate way to control fishing impacts within an EAM? If the pressure from fishing on ecosystem components such as the benthic invertebrate community can only be modelled using fishing effort data as the input variable, then only if fishing effort is related to TACs through some well understood and well defined relationship can TAC management successfully achieve management objectives for these non-target components of marine ecosystems.

The relationship between TACs and fishing effort has, to our knowledge, not been explicitly examined, although within stock assessment circles it is widely assumed that variation in TAC will in fact have little effect on levels of fishing effort. Some simple fisheries science theory suggests that within a single stock system where the population dynamics of the species concerned are well understood and well represented in the assessment models, where fishermen's behaviour is entirely compliant with the rules under which they are obliged to operate and discarding is limited, and where management objectives are to maintain a constant fishing mortality rate, for example f_{lim} (the mortality rate above which stock biomass is likely to be reduced to a point that will compromise future recruitment) or f_{msy} (the mortality rate consistent with achieving maximum sustainable yield from the stock), levels of fishing effort should be constant over time and independent of both variation in TAC and SSB. A rudimentary assumption in fisheries science is that catch (C) per unit effort (E) is proportional to abundance (N) such that:

$$\frac{C}{E} = qN \tag{1}$$

where q is the catchability coefficient; the proportion of fish in the path of the trawl that is actually retained in the net (King, 1995; Jennings et al., 2001). For any given abundance and catchability coefficient therefore, the effort required to take a certain catch is determined as:

$$E = \frac{C}{qN} \tag{2}$$

Assuming that fishing mortality (f) is maintained at a constant rate, then the catch in each year can be determined by:

$$C = \frac{f}{f + m} N\left(1 - e^{-(f + m)}\right) \tag{3}$$

where m is the rate of natural mortality (Jennings et al., 2001). Substituting Eq. (3) into Eq. (2), the catch term disappears altogether indicating that effort is not related to catch and the abundance terms cancel leaving effort simply related to the three terms, f, m and q, which for the sake of this example, these are assumed to be constant.

$$E = \frac{\frac{f}{f + m}\left(1 - e^{-(f + m)}\right)}{q} \tag{4}$$

In reality, particularly within mixed species fisheries where the abundance and TAC of any one stock varies independently of all other stocks being targeted in the same region, resulting in variable discarding, illegal landing and mis-reporting practices, effort might well vary from year to year as fishermen respond in a variable manner to the changing restrictions under which they operate (Biais, 1995). But the essential conclusion remains the same; there is no underlying basis for expecting levels of fishing activity to vary directly in relation to changes in the TACs that are set each year.

If indeed levels of fishing effort are not related to variation in TACs, then this has important implications for fisheries management in the future and the implementation of an EAM. It may mean that catch limitation through the imposition of TACs has no future. If there is no predictable relationship between TAC response indicators and fishing effort based activity indicators, then TAC response indicators cannot be expected to influence pressure indicators in such a way as to achieve desired goals for state indicators of non-target components of the marine ecosystem.

In this paper therefore, we examine explicitly the relationships between TACs for the major demersal fish stocks in the North Sea (cod, haddock, whiting, saithe, plaice and sole) and fishing effort by beam and otter trawlers, the two types of fishing gear predominantly used to target these species. Firstly we examine the relationship over time between TAC and fishing effort by national fleets at the scale of the whole North Sea, focusing on the relationships for particular species and the specific gears used to target them. Secondly we examine the same temporal relationships, but at the scale of the ICES rectangle, to determine the extent to which any relationships between TAC and effort might vary across space.

2. Material and methods

Fishing effort and TAC data for the North Sea (ICES management area IV) were compiled during the EU-project MAFCONS for the period 1997–2004. Fishing effort data, recorded as hours fishing, were available for the German, Norwegian and English fleets. Effort for the