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The Economics of European Eel Management

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The Economics of European Eel Management

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Abstract
The European eel stock is endangered. The European Union has, therefore, introduced strict policies to try to reverse the eel's decline and reduce the threats to its survival. However, the European Union's eel management policy has been implemented on a 'one-size-fits-all' basis, where all the affected countries have been given nearly identical targets, regardless of either the individual country's costs for reducing damages to eels or its importance for the overall eel stock. In this paper, we draw on data from the different national eel management plans as well as from independent studies to compare the cost of measures to reduce eel mortality imposed in different countries. We compare the overall costs to those that could have been incurred with a union-wide, rather than fragmented, abatement program, and find that such a comprehensive management program would have been substantially cheaper and would have affected poorer member countries less.

1. INTRODUCTION

In this paper, we study the European Council (EC) Regulation No. 1100/2007 on eel management and discuss its implementation from an economic efficiency point of view. This efficiency will be measured in terms of the marginal cost of reducing eel mortality for each country as well as the overall costs for the European Union (EU) as a whole. We find that there is considerable scope to improve the efficiency of European eel-protection policies.

Since the 1970s, the stock of the European eel, *Anguilla anguilla*, has declined dramatically due to mortality factors caused by humans, such as overharvesting. Since 2012, the European eel has been classified as critically endangered on the International Union for Conservation of Nature (IUCN) *Red List of Threatened Species*. Traditionally, actions to save the eels have been taken at the local and national level, but in 2007, the European Council established measures in EC Regulation No. 1100/2007 that aimed to organize the recovery of the stock across Europe. The Regulation requires each member state with natural habitats for the European eel to create an eel management plan (EMP), with the purpose not only of reducing fishing mortalities, but also of increasing the escapement levels of this species at its so-called silver eel phase to marine waters (Freyhof and Kottelat 2010; European Council 2007). The Regulation sets two different targets for each member state affected by it: firstly, that national EMPs should include measures to ensure a level of escapement to sea of silver eel biomass equal to at least 40% of the estimated pristine escapement levels; and secondly, that either the catch or the fishing effort of the national eel fishery should be reduced by at least 50% compared with the average level between 2004 and 2006 (European Council 2007). Thus, although the Regulation was determined at EU level, its implementation follows the normal EU approach of setting targets for individual member states.

We begin the paper by discussing the current state of the European eel and the status of current European eel policy. This is followed by a description of the data from the various EMPs used for our analysis. Our analysis consists of descriptive statistics over estimated cost of various measures implemented, and a comparison of these costs with those that would be incurred in a hypothetical joint management scenario. We conclude by discussing the implications for future European eel policy.

2. THE EUROPEAN EEL

Anguilla anguilla lives in ocean waters during the spawning phase of its life cycle, but in freshwater and brackish coastal water during its maturation. Habitats for the species in its maturation phase are found in the North and Baltic Seas, in the Mediterranean and rivers feeding into it, and in parts of Asia. The species is also found as far south in the Atlantic as the Canary Islands (Freyhof and Kottelat 2010). The European eel is sometimes found in the Black Sea, but this area is not considered a natural habitat for the species (European Commission 2009).

Little is known with complete certainty about the life cycle of the European eel, but it is believed that the spawning occurs between March and July in the Sargasso Sea and that, after spawning, the adult eels die. The larvae, also called *leptocephali*, are then transported by the Gulf Stream and the North Atlantic Drift to the European and North African coasts (Kirkegaard 2010). The larval stage of the eel's life cycle lasts around one or two years, after which the larvae metamorphose into glass eels. They then mature into yellow eels, which begin to migrate upstream into river systems and settle in freshwater bodies. They remain there until they mature into sexually mature so-called silver eels, at which point they begin their migration back to the Sargasso Sea to spawn. However, the duration of the transition from glass eel to yellow eel to silver eel differs substantially: the generation length is frequently several decades and depends on variables such as ecosystem characteristics and the temperature of the habitats. The eels in southern Europe generally have shorter life cycles than the eels in northern Europe (Curd 2010; Freyhof and Kottelat 2010).

The long potential lifespan of the eel increases its risk of being killed by predators or artificially instead of dying of old age. Since the eel only spawns at the end of its lifespan, anthropogenic factors that kill it before then can have an important impact on recruitment levels and thereby on the future stock. Among the anthropogenic factors to which the European eel stock is exposed are hydropower turbines, which kill migrating eels, and eel harvesting by commercial or recreational fishermen. In 2009, the International Council for the Exploration of the Sea (ICES) declared that the stock level of the European eel was at a historical low. Furthermore, the recruitment levels of the stock were also historically down and were predicted to continue their negative trend, reaching only between 1% and 9% of the recruitment levels seen in the 1970s (Kirkegaard 2010). Due to the rapid decline of the stock and the problems occurring because of the long generation cycles, as mentioned earlier, the European eel is on the *Red List* as critically endangered.

2.1 EC Regulation No. 1100/2007

In order to reduce the mortality of the European eel and diminish the risk of its possible extinction, the European Council implemented EC Regulation No. 1100/2007. As mentioned in the Introduction section above, this Regulation required all EU member states with natural habitats for European eel to establish a national EMP before July 1, 2009. One of the objectives of such EMPs is to ensure an escapement level into ocean waters of silver eel biomass equal to 40% of the estimated pristine escapement levels (European Council 2007). The stated Regulation does not, however, declare a time frame within which this particular objective should be met; in some national EMPs, the applicable time frame spans up to 200 years. Long time horizons like these can be explained by the relative length of each phase in the eel's life cycle. Nonetheless, the goal of a 40% escapement level can be reached by some areas or countries in the short term (ICES Secretariat 2010).

Furthermore, the Regulation in question also required EMPs to include measures to reduce the fishing effort, or the total catch, by 50% in comparison with the average value for the period 2004–2006. Action by EU member states to meet this goal was to have been initiated by January 1, 2009 at the latest (European Council 2007). The Regulation also states that 60% of eels less than 12 cm in length should be reserved for restocking; the Regulation required this goal to be met by no later than July 31, 2013 (Freyhof and Kottelat 2010; European Council 2007). Countries within the EU that have no natural habitats for the European eel were to apply by January 1, 2008 for exemption from the obligation to draft a national EMP. The European Commission exempted any member state that could show that their inland or marine waters did not constitute natural habitats for this species (European Council 2007).

3. OVEREXPLOITATION OF THE EUROPEAN EEL

As a single, ocean-based breeding population, the European eel can be considered a common-pool resource on both a national and an EU level (see e.g. Hoffmann and Quaas 2016 for a discussion of European fisheries policy in general in this light). This brings additional complexity to the common-property management problem, since the actors within a nation can be assumed to act rationally and aim to maximize their benefits, as well as the countries themselves. However, the incentives for an individual country to implement measures that restrict actors within the nation may well be limited, since the effect of such measures will mostly benefit other countries. Therefore, there is a risk that individual countries will do as little as possible to reduce their own costs for protecting the eel; this will in turn lead to a situation where the overall effort is too limited. In addition to this general problem, in the case of the European eel, the countries downstream along the eel's migration route will benefit from countries upstream improving their eel management, but the latter countries have no incentive to take this into consideration. Since the creation of the Regulation in question, free-riding behavior has presumably been reduced to some extent—not only because of the increased requirement for information-sharing, but also because the EU is able to act as an enforcing authority. However, since the Regulation does not specify any consequences for countries not adhering to its stipulations, the risk is that enforcement levels may be low.

A common means of implementing EU policy is to reach EU-wide goals through decentralized national plans that aim to achieve preset national targets, rather than through a single, EU-level plan aimed at a joint target (see e.g. Weale 1999 for a discussion of EU environmental policy). This implies limited coordination between different countries' policies and limited use of synergies between such policies, which in turn entails that policies may become unnecessarily costly. On the other hand, the advantage of national-level policies is that they can be tailored to local circumstances and can exploit knowledge of local conditions more effectively than a single, EU-level policy could. Indeed, in many cases, the efficiency gains from localized policies are greater than the efficiency losses caused by a lack of EU-wide coordination. Moreover, local implementation is frequently combined with targets that are tailored to local conditions, and this provides scope for the EU to take into consideration concerns other than efficiency, such as equity, when targets are set. Aside from potential efficiency gains, this local implementation theoretically increases local ownership of the policies in question.

However, the eel regulation explicitly sets identical national targets for all member countries and specifies a limited number of measures (three in all, discussed below) that can be carried out to achieve these targets in the short term.

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Moreover, unlike many other environmental issues where the problems are more localized, in the case of the eel, the problem is that of managing a joint European biological resource which is common to all the participating countries. The efficiency losses from limited coordination between different countries can therefore be expected to be greater for eel management than for many other environmental issues. It is therefore worthwhile to compare the cost of implementing national EMPs with that of implementing a joint plan.

4. DATA

Since EC Regulation No. 1100/2007 has not yet been evaluated and there is no information on its performance, the data used in this study have been collected from multiple sources. The countries included in the study were chosen based on the availability of national EMPs published in English. In cases when a national EMP could only be found published in the national language, the country concerned was included in the study only if other sources presented a summary or explanation of general measures in its EMP in English.

The data on measures implemented by member states were collected from the 2009 national EMPs. For the United Kingdom (UK) and Sweden, official documents published by the relevant national government agencies were used to determine the economic cost of the national measures implemented. For all of the other countries included in the study, these costs were estimated after analyzing various national data as well as information published by the ICES and other external sources. The assumptions and calculations made for specific countries have in some cases been applied generally to all of the countries in the study. In addition, many of the calculations for national measures implemented are based on guesstimates rather than solid statistical information.

The estimated pristine escapement levels of eel used in this study were collected from the various national EMPs. In some cases when such estimations were not available in the EMP, ICES estimates were used (ICES 2012a, 2012b, 2012c). However, both member states and the ICES acknowledge that the pristine escapement levels are very hard to determine. As a result, since an individual country has a certain degree of incentive to minimize its own efforts, there is a risk that countries may understate estimates of their pristine stock levels (Bevacqua et al. 2009; ICES 2012a, 2012b, 2012c; Belgium 2009). Moreover, the methods used to determine the pristine escapement levels vary among the countries, but the one they principally employ is to assume that the catch is proportional to the population size. Another popular alternative is to calculate the estimated level of eel density based on the area of eel habitats in a country (Sweden-2008). Both these methods are permitted by the Regulation.

Since return migration is measured in returning silver eels per year, costs are recalculated into annualized present value costs; this allows for an easy comparison of costs per year with eels per year. However, it should be noted that the annualized present value costs of each measure are calculated as from the start of the current year, whereas the measures will, in most cases, only affect return migration after some delay. For purposes of present value calculations, a 3% discount rate is assumed.

4.1 The Measures

The measures found in the various national EMPs can be divided into three categories. These are: (1) restrictions on eel fishing, (2) reduction of hydropower-related mortalities and migration barriers, and (3) stocking. These measures are described in more detail below.

4.1.1 Restrictions on Eel Fishing

Restrictions on eel fishing consist of several national measures that all aim to reduce eel catches or eel-fishing efforts. One example of these measures is to restrict the fishing period by reducing the number of days when eel fishing is allowed (The Netherlands 2008). This type of measure ranges from a two-week reduction in the eel-fishing season to a total ban on all eel-fishing activities. Another national measure is to reduce the number of actors by introducing licensing requirements for commercial fishing (Belgium 2009; Sweden 2008). However, the practical impact of such measures is debatable. In other fisheries, experience has shown that with time, effort tends to intensify during the remaining fishing season or among the remaining fishermen. Thus, the overall reduction in fishing becomes smaller than expected, unless fishing is banned completely (estimates reported in Stage 2015, for example, suggest that such an increase in effort has already happened in the Swedish eel fishery). Yet, for the purposes of this study, we assumed that the reduction in eel-fishing efforts has the intended effect of reducing eel mortalities.

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This study also assumed that all measures aiming to reduce fishing activities have a zero economic cost, meaning that—due to rent dissipation and low economic efficiency within the sector—society does not suffer any economic loss if eel fishing is restricted. This assumption is based on estimates and calculations made in numerous previous studies of the profitability of European fishing (see e.g. Schroeer et al. 2011, or Swedish EPA 2009). For example, when catches from open-access fisheries such as those for the European eel are reduced, the economic losses are usually zero. Because of this assumption, all of the fishing measures carried out by an individual country are regarded as a single national measure. Regardless of how many eels are saved by this measure, the marginal abatement cost for each country will still be equal to zero. However, as part of our sensitivity analysis, we include the possibility that the harvested eels have a social value as consumption goods equal to 10% of their unprocessed market price, i.e., approximately 65 Euro cents per eel. In addition, reduced fishing of silver eels is assumed to affect return migration immediately, while reduced fishing of glass and yellow eels only affects the return migration of silver eels with a time lag corresponding to the time remaining to complete maturation.

4.1.2 Reduction of Hydropower-related Mortalities and Migration Barriers

One major reason for the high mortality levels of the European eel is the large number of eels that are injured or killed when they attempt to pass hydropower turbines and stations. Estimations made by the ICES (2003) indicate that on average, almost 30% of the eels passing through a hydropower station are damaged or killed. This is a major problem, since many eels have to pass several hydropower stations during migration. In some areas, the mortality levels attributable to this cause are so high that no migrating eels presumably reach marine waters, i.e., the mortality level is assumed to be 100% (Czech Republic 2008).

The mortality rate can be reduced if eel passes are installed at the relevant hydropower plants. There are many different types of eel passes, and new types are rapidly being developed. These passes can be installed for both big and small hydropower stations, as well as in areas where there are other migration barriers. For many of the sites, there is a need to install passes for both upstream and downstream migration and sometimes more than one in each direction, making the installations expensive (ACE 2012; Bakken 2011; Environment Agency 2011).

In this paper, for simplicity's sake, all the member states in the study were assumed to use similar eel passes. It was also assumed that the cost of an eel pass was approximately equal in all of the countries concerned, and that the average annual cost of maintaining the eel pass and of lost electricity production was equal to approximately one quarter of the initial investment cost. Most countries' EMPs did not specify the costs for installing eel passes, so the cost estimates made for the UK were used for all of the countries in the study (United Kingdom 2009). Of the three types of measure studied in the paper, the cost of hydropower-related measures was the most complicated to calculate. For the UK and Sweden at least, the study could use the costs estimated for this measure in those two countries' national reports (Swedish EPA 2009; United Kingdom 2009). In other countries, such as Belgium, the national EMPs only calculate the number of eel passes that need to be built; these calculations were then combined with the UK cost estimate per pass to arrive at cost estimates for those other countries. Where countries did not state the number of eel passes that needed to be built, it was assumed (based on estimates from Belgium 2008) that passes would be built in 60% of their hydropower station sites.

For all countries, we further assumed that installing eel passes would ensure that the species could enjoy as safe a passage as it had before the hydropower plant was built. If such passes are not 100% effective, as assumed here, the cost per return-migrating silver eel will be higher. As part of our sensitivity analysis, we include the possibility that the compound effect of having to negotiate several imperfect eel passes reduces the number of surviving eels by half, thus doubling the cost per return-migrating silver eel. The downstream migration of silver eels is assumed to improve as soon as the eel passes are completed—a process that is assumed to take five years. However, the length of time needed for the yellow eel to mature into its silver eel phase means that an improved upstream migration of yellow eels only affects the return migration of silver eels with a lag corresponding to the length of the maturation stage.

4.1.3 Stocking

The third measure used in eel management is stocking. *Stocking* comprises moving glass eels or juvenile eels from areas of high eel density (chiefly the Bay of Biscay) to areas of low density. Historically, this measure was taken to improve the quantities of eel available for fishing upstream of hydropower plants that have reduced the naturally available eel numbers; however, after the implementation of the relevant Regulation, this type of stocking has been heavily reduced. Most EU member states are increasingly aware of the issue of turbine mortality and, therefore, now primarily stock eels in areas where there are open migration routes to marine waters (Czech Republic 2008; The Netherlands 2008; Sweden 2008).

The ICES has flagged a number of problems that could arise from large-scale stocking. They state that stocking measures might affect the eels' migration behavior and that relocating eels in this fashion may entail that they are unable to find their way back to the Sargasso. There is also the risk of spreading viruses to previously unaffected subpopulations (ICES Secretariat 2010). It is still uncertain whether such stocking is effective in regard to its contribution to meet the target of increasing the escapement of silver eel biomass to sea to 40% of the pristine escapement levels. However, we assume here that the eels that are stocked are in fact able to maintain their natural migration patterns, and are therefore able to contribute to spawning when they reach the Sargasso.

Yet even if this assumption is correct, stocking will not necessarily contribute to increasing overall spawning, since the management measure merely entails moving eels from one part of the overall eel stock to another. Stocking may have been justifiable historically when eel density and hence natural mortality were extremely high in the source waters in the Bay of Biscay, such that moving eels from the Bay to lower-density waters improved their chance of survival. With the current depleted state of the stock across Europe, however, eel density and natural eel mortality are low even in the Bay, and moving eels from the Bay to other locations may not improve their chance of contributing to overall spawning. The ICES, therefore, discourages stocking and recommends it only when clear scientific evidence suggests that stocking in a specific area will improve the overall status of the spawning stock—evidence that is currently not available for any of the relevant areas (ICES Secretariat 2010). An outright ban on fishing of glass eels and hence on stocking would lead to worse eel remigration outcomes for those countries that would no longer be able to stock their waters with glass eels from the Bay. However, it would lead to an increase of the number of eels remaining in the Bay and, given the above discussion, it seems likely that this would improve the overall viability of the eel spawning stock. We examine this policy option accordingly later in the paper.

Eel mortality rates differ from country to country, as well as mortality estimates for stocked eels. Thus, although the cost for stocking 1 kg of eel (e.g., as estimated in Swedish EPA 2009) can be assumed to be roughly the same everywhere, the cost per silver eel actually expected to be added to the migratory stock varies among the countries. It depends not only on the age of the eels at stocking, but also on their expected subsequent mortality rates. National EMPs sometimes state the number of eels they plan to stock, yet for some of the countries included in this study, no such estimates are provided. For the latter countries, average stocking levels for the period 2000–2010 have been used as estimates (ICES Secretariat 2010).

5. ANALYSIS

Reduced fishing is estimated to account for over half of the total increase in return migrations (see Table 1). As noted in subsection 4.1.1 above, the cost of this measure is assumed to be zero for all affected fisheries, but it remains one of the cheapest options—even in the sensitivity analysis, where an estimated cost is introduced. Reduced hydropower mortality accounts for an uncertain additional increase in return migrations, with widely differing annualized present value costs, i.e., ranging from 2 cents (Denmark) to 59 Euros (Latvia) per eel saved. Stocking measures also vary widely in cost, depending on estimated survival rates; for example, stocking costs 55 cents per additional statistical silver eel in France (where stocking essentially entails catching glass eels in one part of French waters and releasing them somewhere else in the country), but 185 Euros in Estonia (where stocked eels need to be transported longer distances before being released and where they face higher mortality rates after being released). The overall time profile of the impacts is slanted well towards the future, with only some 20% of the anticipated increase in return migration occurring in the first decade, and around two thirds of the overall impact delayed by 30 years or more. This multi-decade delay before affecting return migration applies not only to almost all stocking (since it is primarily done using glass eels, which take decades to mature), but also to measures that lead to improved upstream migration.

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Table 1. Comparison of Principal Eel Management Measures for the Region Studied.

Measure	Total cost, Annualized Present Value (EUR)	Increase in silver eel return migration				Cost per eel, Annualized Present Value (EUR)
		1st decade	2nd decade	4th decade onwards	Total	
Restrictions on eel fishing	0–7.7 million	5.4 million	1.5 million	14.4 million	21.2 million	0.00–0.65
Reduction of hydropower-related mortalities and migration barriers	12.2 million	1.8–3.6 million		1.8–3.6 million	3.6–7.3 million	0.02–59
Stocking	31.5 million		0.6 million	6.3 million	6.8 million	0.55–185
Overall	43.8–51.5 million	7.2–9.0 million	2.0 million	22.5–24.3 million	31.7–35.3 million	0.00–185

Authors' estimates are based on Belgium (2008); Czech Republic (2008); Denmark (2008); Estonia (2008); European Commission (2009); Helsinki Commission Sweden (2010); ICES (2012a, 2012b, 2012c); ICES Secretariat (2010); Latvia (2008); Poland (2008); Punys and Pelikan (2007); Sweden (2008); Swedish EPA (2009); The Netherlands (2008); and United Kingdom (2009). There is no increase in return migration in the third decade. Components may not sum exactly to the displayed totals due to rounding.

The countries for which the marginal costs are lowest incur present value costs of less than 1 Euro per eel even for their most expensive measures, while the country with the highest marginal cost, Estonia, incurs a cost of 185 Euros per eel for its most expensive measures. The large differences in costs among the various countries mean it would be valuable to examine how much less expensive a least-cost approach would be.

Let us assume firstly that, instead of the current set of national EMPs, the EU (1) reduced eel fishing to zero in those countries that currently fish the most, i.e., France, Portugal, Spain, and The Netherlands, (2) retained its current fishing reductions in other member countries, and (3) banned stocking, since this merely entails moving eels from one part of the stock to another with minimal benefit. Secondly, let us assume that eel survival rates are at least as high in their original habitat as in the locations where they are currently being stocked as part of the various national EMPs. The overall costs of achieving a slightly bigger increase in eel remigration than is currently the case—and with a better time profile (see Table 2)—could then be around 73–100% lower, depending on the cost of the lost fishing. Thus, the current combination of national plans is not cost-effective. Provided that the current management system is socially desirable (that is, the existence value from maintaining a viable eel stock is high enough to justify the current costs), then achieving better survival rates faster and at lower costs would presumably be preferable.

Table 2. Estimated Cost of Achieving the Same Mortality Reduction with a Ban on Fishing in Key Countries and a Simultaneous Ban on Stocking.

Measure	**Total cost, Annualized Present Value (EUR)**	**Increase in silver eel return migration**				**Cost per eel, Annualized Present Value (EUR)**
		1st decade	**2nd decade**	**4th decade onwards**	**Total**	
Restrictions on eel fishing	0–13.9 million	9.1 million	2.4 million	28.6 million	40.2 million	0.00–0.65
Overall	0–13.9 million	9.1 million	2.4 million	28.6 million	40.2 million	0.00–0.65

Authors' calculations. There is no increase in return migration in the third decade. Components may not sum exactly to the displayed totals due to rounding.

The EU comprises member countries with widely varying income levels, and equity concerns frequently trump efficiency concerns when more cost-effective policies would have unacceptable distributional implications. Thus, one could imagine that the current implementation of eel conservation, with high costs in some countries and low costs in others, might be justifiable on equity grounds. This is because the higher costs affect relatively well-off countries, while less prosperous countries have lower conservation costs. However, a simple correlation analysis indicates that the (weak) statistical correlation between different countries' marginal cost and their per-capita gross domestic product is in fact negative (i.e., on average, poorer countries pay higher marginal costs, and those costs fall with increasing average income).

6. DISCUSSION

This paper reports on a study of the costs of implementing European eel conservation as a collection of national EMPs rather than as a joint, EU-wide policy. We make assumptions about the biological impacts of existing fishing and stocking policies that are more optimistic than those made by many biologists. Despite this, we find that even a very simplistic policy of banning eel fishing in the most important fishing countries could reduce the implementation costs dramatically, compared with the current set-up. A more coherent policy, where synergies between different measures are taken into account more fully, could presumably reduce overall costs even further. Even though the countries that would be most affected by such an alternative plan would probably demand compensation from other member countries, the cost savings compared to the current set-up should be enough to pay for such a compensation scheme. Other than the cost of setting up the compensation mechanism, there would not necessarily be any additional costs linked to a more coherent policy; monitoring an outright ban on eel fishing, in those countries where such a ban would be introduced, should not be costlier than monitoring reduced fishing.

The EU has a long tradition of carrying out joint policies by setting national targets and having policies implemented at the national level, rather than the EU-wide level. In many cases, this is probably justifiable, because the gains from using local knowledge in local policy implementation are greater than the efficiency gains of a joint policy. However, when the policy problem is truly a joint one—as in the case of the eel, where all EU countries are damaging the same eel stock, then the efficiency losses from national implementation risk becoming dramatic. The current assumption, that local implementation is preferable by default, no longer holds when such a large share of the gains from the implementation accrue elsewhere. There appears to be a need not just for joint target-setting, but also for joint implementation.

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