Towards sustainable management of huntable migratory waterbirds in Europe

A report by the Waterbird Harvest Specialist Group of Wetlands International

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Executive Summary

The EU Birds Directive and the African-Eurasian Waterbird Agreement provide an adequate legal framework for sustainable management of migratory waterbird populations. The main shortcoming of both instruments is that it leaves harvest decisions of a shared resource to individual Member States and Contracting Parties without providing a shared information base and mechanism to assess the impact of harvest and coordinate actions in relation to mutually agreed objectives.

A recent update of the conservation status of waterbirds in the EU shows that almost half of the populations of species listed on Annex II of the Birds Directive have a declining short-term trend and over half of them are listed in Columns A and B of AEWA. This implies that their hunting could either only continue under the framework of an adaptive harvest management plan or their hunting should be regulated with the view of restoring them in favourable conservation status.

We argue that a structured approach to decision-making (such as adaptive management) is needed, supported with adequate organisational structures at flyway scale. We review the experience with such an approach in North America and assess the applicability of a similar approach in the European context. We show there is no technical reason why adaptive harvest management could be not applied in the EU or even AEWA context.

We demonstrate that an informed approach to setting allowable harvests does not require detailed demographic information. Essential to the process, however, are estimates of either the observed growth rate from a monitoring program or the growth rate expected under ideal conditions. In addition, periodic estimates of population size are needed, as well as either empirical information or reasonable assumptions about the form of density dependence. We show that such information exists for many populations, but improvements are needed to improve geographic coverage, reliability and timely data availability.

We highlight the importance of the International Waterbird Census and specialised goose and seaduck monitoring in estimating population sizes and observed growth rate of the populations. We encourage further investments into the development of these schemes. We also recognise the importance of migration studies to improve our understanding of delineations of populations.

We also highlight that, with a few exceptions, the available data does not allow the European Commission, competent authorities of the Members States or other AEWA Contracting Parties to assess levels of harvest and their sustainability and, therefore, regulate hunting accordingly. Therefore, we recommend that annual reporting on harvest levels of waterbird populations would be gradually introduced in the EU and the AEWA region.

We propose that future AEWA and EU action plans and management plans for Annex II species should apply the principles of adaptive harvest management framework and make provisions for setting up adequate monitoring and information management systems and organisational structures to manage the decision-making process. We suggest that internationally coordinated management structures are established to facilitate dialogue, learning and communication between stakeholders with different interests and cultural backgrounds.

Policy Background to Sustainable Harvest Management in Europe and in a Flyway Context

The need for international coordination in the management of migratory huntable waterbird populations has been recognised for a long time. Globally, the need to protect wetlands as habitats for migratory waterbirds was first enshrined in the Ramsar Convention on Wetlands (1971). Additionally, the Convention on Migratory Species (CMS) recognised the need for measures other than just habitat conservation. This broader approach was also followed by the Directive 2009/147/EC of the European Parliament and of the Council on the conservation of wild birds (better known as the Birds Directive). The majority of EU Member States are also Contracting Parties to the African-Eurasian Waterbird Agreement (AEWA) that provides a management framework for the entire flyway including countries outside of the European Union.

According to the requirements of the Birds Directive, Member States of the European Union shall maintain the populations of European bird species at a level that corresponds to ecological, scientific and cultural requirements, while taking account of economic and recreational requirements or to adapt a population to that level. Such requirements, amongst others, include hunting as one of the legitimate uses of waterbirds and also recognising the positive effects hunting can have on waterbird populations through habitat maintenance and predator control. The Directive tries to achieve the above mentioned objective through habitat conservation and through provisions that regulate the disturbance and utilization of European bird species, including by hunting. In principle, only the species listed on Annex II of the Directive can be hunted across the EU or in certain Member States, but in all cases Member States shall ensure that the hunting of these species does not jeopardise conservation efforts in their distribution area (range). The 'Guidance Document on Hunting under Council Directive 79/409/EEC on the conservation of wild birds' (European Commission 2008) provides interpretation of the requirements of the Birds Directive. However, neither the Birds Directive nor the Hunting Guide intend to provide a mechanism for coordinated, science-based harvest management to address the sustainable management of populations of waterbirds, be they threatened, in favourable conservation status or conflicting / overabundant species. The Hunting Guide recognises that, "so that hunting does not lead to the decline of huntable species the general approach in wildlife management is to ensure that hunting of species does not exceed the range between 'maximum' and 'optimum' sustainable yield" (European Commission 2008). However, the Hunting Guide does not provide any specific guidelines to the national governments about how this can be achieved in the case of migratory species other than the avoidance of high levels of exploitation. Therefore, it is clear that a better understanding is needed of the flyway-level impact of harvesting and the state-of-the-art management principles, in order to assist Member States in meeting the requirements of the Habitat and Bird Directives.

AEWA came into force in 1999 under the framework of the CMS. It provides a management framework for the entire flyway of 555 populations of 255 migratory waterbird species according to their global conservation status (i.e. their listing on Appendix I of CMS and on the IUCN Red List), their population size, distribution and population trend. In general, Contracting Parties shall maintain or restore migratory waterbird populations to favourable conservation status. To this end, they shall prohibit the taking of birds and eggs of populations listed in Column A of Table 1 to the AEWA Action Plan. Hunting of populations listed in Categories 2, 3 and 4 in Column A can be conducted only within a sustainable use framework, ideally following the principles of adaptive harvest management. In the case of populations listed on Column B, Contracting Parties should regulate harvest with the view to maintaining or contributing to the restoration of those populations to a favourable conservation status and to ensure that any harvest is sustainable. Amongst other means this includes establishing harvest limits, where appropriate, and providing adequate controls to ensure that these limits are observed. Recently, AEWA started the implementation of internationally coordinated harvest management in the case of the Svalbard pink-footed goose *Anser brachyrhynchus* (Madsen & Williams 2012), and a process is underway for the taiga bean goose *Anser fabalis fabalis*. These management planning processes provide important lessons for the coordinated harvest management of a wider range of migratory waterbird species and may help to expand coordinated harvest management.

In this report we present the current status of huntable waterbirds within the EU and we lay out the basic principles of sustainable harvest that are the prerequisite for their exploitation. Combined with an overview of the current state of internationally coordinated monitoring of populations and their demography, as well as harvest, we provide recommendations on how to improve EU harvest management protocols and procedures on a flyway basis.

This report is timely for a number of reasons. Firstly, many waterbird populations listed on Annex II of the Birds Directive are declining in the European Union. According to AEWA and the Birds Directive, their continued hunting would require the preparation of management plans that would help them return to favourable conservation status. This is consistent with the recognition that continued hunting may provide incentives for habitat conservation measures that benefit the target species and where a hunting ban might therefore be counterproductive. However, the EU management plans that have been produced so far did not provide any framework for coordinated harvest management within the EU or at flyway level and thus it cannot guarantee that continued harvests have not jeopardised conservation efforts elsewhere in the flyway.

Secondly, populations of a number of waterbird species, both huntable and non-huntable, are rapidly increasing, and some of them cause damage or concern to agricultural crops, fishery interests, air safety and biodiversity. However, even in the case of these populations, ecological, scientific, cultural, economic and recreational requirements should be coherent, not only at the level of individual Member States but also at the flyway scale, yet this is not currently the case. In 2015, the European Commission will publish a report on the State of Nature in the EU with results from the new Article 12 reporting format, which for the first time contains quantitative information on bird populations in the EU. This will provide a basis for better implementation of the Directive and a means to measure progress towards target 1 of the EU Biodiversity Strategy.

Thirdly, the European Commission has launched a fitness check of the Birds Directive under its Regulatory Fitness and Performance Programme, to be conducted in 2015. Although this report will show that the Birds Directive together with AEWA provide an adequate legal framework for sustainable harvest management at the flyway scale, we highlight the need for improving the generation of knowledge concerning the status and utilisation of waterbird populations and creating adequate international coordination mechanisms to ensure the sustainable harvest of migratory waterbirds at flyway scale. It is our hope that these recommendations can provide useful input to the EU fitness check process.

Status of Huntable Waterbirds in the European Union

Annex II of the Birds Directive lists 50 waterbird species in total, of which 16 are listed in Part A, i.e. they can be hunted across the European Union. The remainder (Part B) can only be hunted in certain Member States. These 50 species correspond to 83 biogeographic populations on AEWA's Table 1[1]. From these 83 populations, 30 are listed in Part A and 53 in Part B of Annex II.

Out of the populations listed in Part A of Annex II of the Birds Directive, one population, the taiga bean goose, is listed in Column A Category 3c* in Table 1 of the AEWA Action Plan. This means that the hunting of this population can continue only under the framework of an adaptive harvest management plan (that is currently being prepared under the framework of AEWA). Another 13 populations are listed in Column B of Table 1 of the AEWA Action Plan, i.e. their harvest should be regulated with the view to restoring them to or maintaining them in favourable conservation status. In Part B, however, there are six populations listed in Column A, including three populations of Globally Near Threatened Species and two Globally Threatened Species, the long-tailed duck *Clangula hyemalis* and velvet scoter *Melanitta fusca*. In addition, 22 populations from Part B of Annex II are listed in Column B of AEWA.

[1] Only those populations considered here that can be hunted in at least one Member State. Populations overlapping marginally only with Cyprus are excluded.

Considering the short-term (10-years) trends, of the 83 Annex II populations 35 (42%) have declined over the last decade, 27 (32%) were stable, fluctuating or uncertain and 21 (25%) have increased. Many of the increasing populations are Arctic breeding geese that can cause serious crop damage at their wintering and staging areas, while in some cases their increased populations may also affect sensitive natural vegetation at their breeding grounds. The management of such populations would also require an adaptive management framework to maintain their population at levels that correspond to ecological, scientific and cultural requirements while taking account of economic and recreational requirements.

Table 1. Status of waterbird populations listed in Annex II of the EU Birds Directive, based on Wetlands International (2014).

Birds Directive Annex 2	AEWA Column A	AEWA Column B	AEWA Column C	Species	Population	10-years trend
Part B			1	Mute Swan *Cygnus olor*	North-west Mainland & Central Europe	Increasing
Part A	3 c*			Bean Goose *Anser fabalis*	*fabalis*, North-east Europe/North-west Europe	Declining
Part A			(1)	Bean Goose *Anser fabalis*	*rossicus*, West & Central Siberia/NE & SW Europe	Stable
Part B		2a		Pink-footed Goose *Anser brachyrhynchus*	East Greenland & Iceland/UK	Increasing
Part B		1		Pink-footed Goose *Anser brachyrhynchus*	Svalbard/North-west Europe	Increasing
Part B			1	Greater White-fronted Goose *Anser albifrons*	*albifrons*, NW Siberia & NE Europe/North-west Europe	Increasing
Part B			1	Greater White-fronted Goose *Anser albifrons*	*albifrons*, Western Siberia/Central Europe	Increasing
Part B			1	Greater White-fronted Goose *Anser albifrons*	*albifrons*, Western Siberia/Black Sea & Turkey	Unknown
Part B	2 *			Greater White-fronted Goose *Anser albifrons*	*flavirostris*, Greenland/Ireland & UK	Declining
Part A		1		Greylag Goose *Anser anser*	*anser*, Iceland/UK & Ireland	Increasing
Part A			1	Greylag Goose *Anser anser*	*anser*, NW Europe/South-west Europe	Increasing
Part A		1		Greylag Goose *Anser anser*	*anser*, Central Europe/North Africa	Increasing
Part A		1		Greylag Goose *Anser anser*	*rubirostris*, Black Sea & Turkey	Unknown
Part B		2b		Brent Goose *Branta bernicla*	*bernicla*, Western Siberia/Western Europe	Declining
Part B	1 c			Brent Goose *Branta bernicla*	*hrota*, Svalbard/Denmark & UK	Increasing
Part A			1	Eurasian Wigeon *Anas penelope*	Western Siberia & NE Europe/NW Europe	Declining
Part A			1	Eurasian Wigeon *Anas penelope*	W Siberia & NE Europe/Black Sea & Mediterranean	Stable
Part A		1		Gadwall *Anas strepera*	North-west Europe	Increasing
Part A			1	Gadwall *Anas strepera*	North-east Europe/Black Sea & Mediterranean	Increasing
Part A			1	Common Teal *Anas crecca*	*crecca*, North-west Europe	Fluctuating
Part A			1	Common Teal Anas crecca	*crecca*, W Siberia & NE Europe/Black Sea & Mediterranean	Increasing

Birds Directive Annex 2	AEWA Column			Species	Population	10-years trend
	A	B	C			
Part A			1	Mallard *Anas platyrhynchos*	*platyrhynchos*, North-west Europe	Stable
Part A			1	Mallard *Anas platyrhynchos*	*platyrhynchos*, Northern Europe/West Mediterranean	Increasing
Part A		2c		Mallard *Anas platyrhynchos*	*platyrhynchos*, Eastern Europe/Black Sea & East Mediterranean	Declining
Part A		1		Northern Pintail *Anas acuta*	North-west Europe	Declining
Part A			1	Northern Pintail *Anas acuta*	W Siberia NE & E Europe/S Europe & West Africa	Fluctuating
Part A		2c		Garganey *Anas querquedula*	Western Siberia & Europe/West Africa	Stable?
Part A		1		Northern Shoveler *Anas clypeata*	North-west & Central Europe (win)	Declining
Part A			1	Northern Shoveler *Anas clypeata*	W Siberia NE & E Europe/S Europe & West Africa	Fluctuating
Part B		1		Red-crested Pochard *Netta rufina*	South-west & Central Europe/West Mediterranean	Increasing
Part A		2c		Common Pochard *Aythya farina*	North-east Europe/North-west Europe	Declining
Part A		2c		Common Pochard *Aythya farina*	Central & NE Europe/Black Sea & Mediterranean	Declining
Part A			1	Tufted Duck *Aythya fuligula*	North-west Europe (win)	Declining?
Part A		2c		Tufted Duck *Aythya fuligula*	Central Europe Black Sea & Mediterranean (win)	Declining
Part B		2c		Greater Scope *Aythya marila*	Northern Europe/Western Europe	Stable?
Part B		2d		Common Eider *Somateria mollissima*	*mollissima*, Baltic Denmark & Netherlands	Declining
Part B			1	Long-tailed Duck *Clangula hyemalis*	Iceland & Greenland	Increasing
Part B		2c		Long-tailed Duck *Clangula hyemalis*	Western Siberia/North Europe	Declining
Part B		2a 2c		Common Scoter *Melanitta nigra*	*nigra*, W Siberia & N Europe/W Europe & NW Africa	Stable?
Part B		2a 2c		Velvet Scoter *Melanitta fusca*	*fusca*, Western Siberia & Northern Europe/NW Europe	Declining?
Part B			1	Common Goldeneye *Bucephala clangula*	*clangula*, North-west & Central Europe (win)	Declining
Part B			1	Common Goldeneye *Bucephala clangula*	*clangula*, North-east Europe/Adriatic	Stable
Part B		1		Common Goldeneye *Bucephala clangula*	Western Siberia & North-east Europe/Black Sea	Increasing
Part B			1	Red-breasted Merganser *Mergus serrator*	North-west & Central Europe (win)	Declining

Birds Directive Annex 2	AEWA Column			Species	Population	10-years trend
	A	B	C			
Part B		1		Goosander *Mergus merganser*	*merganser*, North-west & Central Europe (win)	Declining?
Part B		2c		Water Rail *Rallus aquaticus*	*aquaticus*, Europe & North Africa	Unknown
Part B			1	Common Moorhen *Gallinula chloropus*	*chloropus*, Europe & North Africa	Stable
Part A			1	Eurasian Coot *Fulica atra*	*atra*, North-west Europe (win)	Declining
Part A			1	Eurasian Coot *Fulica atra*	*atra*, Black Sea & Mediterranean (win)	Increasing
Part B		2c		Eurasian Oystercatcher *Haematopus ostralegus*	*ostralegus*, Europe/South & West Europe & NW Africa	Declining
Part B			1	Northern Lapwing *Vanellus vanellus*	Europe/Europe & North Africa	Declining
Part B		2c		Golden Plover *Pluvialis apricaria*	*apricaria*, Britain Ireland Denmark Germany & Baltic (bre)	Declining
Part B			1	Golden Plover *Pluvialis apricaria*	*altifrons*, Iceland & Faroes/East Atlantic coast	Declining?
Part B			1	Golden Plover *Pluvialis apricaria*	*altifrons*, Northern Europe/Western Europe & NW Africa	Stable
Part B			1	Grey Plover *Pluvialis squatarola*	*squatarola*, W Siberia & Canada/W Europe & W Africa	Stable/declining
Part A			1	Eurasian Woodcock *Scolopax rusticola*	Europe/South & West Europe & North Africa	Stable
Part A		2b		Jacksnipe *Lymnocryptes minimus*	Northern Europe/S & W Europe & West Africa	Stable
Part A		2c		Common Snipe *Gallinago gallinago*	*gallinago*, Europe/South & West Europe & NW Africa	Stable
Part A			1	Common Snipe *Gallinago gallinago*	*faroeensis*, Iceland Faroes & Northern Scotland/Ireland	Unknown
Part B	4			Black-tailed Godwit *Limosa limosa*	*limosa*, Western Europe/NW & West Africa	Declining
Part B	4			Black-tailed Godwit *Limosa limosa*	*islandica*, Iceland/Western Europe	Increasing
Part B		2a		Bar-tailed Godwit *Limosa lapponica*	*lapponica*, Northern Europe/Western Europe	Increasing
Part B		2a 2c		Bar-tailed Godwit *Limosa lapponica*	*taymyrensis*, Western Siberia/West & South-west Africa	Declining
Part B			(1)	Whimbrel *Numenius phaeopus*	*phaeopus*, Northern Europe/West Africa	Fluctuating
Part B			(1)	Whimbrel *Numenius phaeopus*	*islandicus*, Iceland Faroes & Scotland/West Africa	Unknown
Part B	4			Eurasian Curlew *Numenius arquata*	*arquata*, Europe/Europe North & West Africa	Declining?

Birds Directive Annex 2	AEWA Column			Species	Population	10-years trend
	A	B	C			
Part B			(1)	Spotted Redshank *Tringa erythropus*	N Europe/Southern Europe North & West Africa	Declining?
Part B			1	Common Redshank *Tringa totanus*	Northern Europe (breeding)	Stable/ Fluctuating
Part B		2c		Common Redshank *Tringa totanus*	Central & East Europe (breeding)	Declining?
Part B			1	Common Redshank *Tringa totanus*	Iceland & Faroes/Western Europe	Declining?
Part B		2c		Common Redshank *Tringa totanus*	Britain & Ireland/Britain Ireland France	Declining
Part B			1	Common Greenshank *Tringa nebularia*	Northern Europe/SW Europe NW & West Africa	Stable?
Part B		2a 2c		Red Knot *Calidris canutus*	Northern Siberia/West & Southern Africa	Declining
Part B		2c		Ruff *Philomachus pugnax*	Northern Europe & Western Siberia/West Africa	Declining
Part B		2c		Mew Gull *Larus canus*	NW & Cent. Europe/Atlantic coast & Mediterranean	Declining?
Part B			1	Mew Gull *Larus canus*	NE Europe & Western Siberia/Black Sea & Caspian	Unknown
Part B			1	Great Black-backed Gull *Larus marinus*	North & West Europe	Increasing
Part B			1	Herring Gull *Larus argentatus*	*argentatus*, North & North-west Europe	Increasing
Part B		2c		Herring Gull *Larus argentatus*	*argenteus*, Iceland & Western Europe	Declining
Part B			1	Caspian Gull *Larus cachinnans*	Black Sea & Western Asia/SW Asia NE Africa	Increasing?
Part B			1	Yellow-legged Gull *Larus michahellis*	Mediterranean Iberia & Morocco	Increasing
Part B	3 c			Lesser Black-backed Gull *Larus fuscus*	*fuscus*, NE Europe/Black Sea SW Asia & Eastern Africa	Declining?
Part B			1	Lesser Black-backed Gull *Larus fuscus*	*intermedius*, S Scandinavia Netherlands Ebro Delta Spain	Increasing

Principles of Sustainable Exploitation

Contribution lead by Fred A. Johnson, Southeast Ecological Science Center, U.S. Geological Survey, Gainesville, Florida, USA

The harvest of renewable natural resources is predicated on the notion of reproductive surplus, and ultimately on the theory of density-dependent population growth (Hilborn et al. 1995). This theory predicts a negative relationship between the intrinsic rate of population growth and population density (i.e. number of individuals per unit of limiting resource) due to intraspecific competition for resources: as a population increases in number of individuals its rate of growth should gradually decline. In a relatively stable environment, unharvested populations thus tend to settle around an equilibrium where births balance deaths. Populations respond to some extent to harvest losses by increasing reproductive output or through decreased natural mortality because more resources are available per individual. Population size eventually settles around a new equilibrium and the harvest, if not too heavy, can be sustained without destroying the breeding stock. Resource managers often attempt to maximize the sustainable harvest by driving population density to a level that maximizes the intrinsic rate of population growth (i.e. the population's level of maximum net productivity) (Beddington and May 1977).

Although the theoretical basis for sustainably harvesting renewable resources is fairly straightforward, the practice of implementation of harvest management has proven to be more challenging. History is replete with cases where uncontrolled variation in harvests or the environment, naive assumptions about system response, or management policies with short time horizons have led to resource collapse (Ludwig et al. 1993). To be successful, sustainable harvesting depends on (i) an ability to effectively regulate the size of the harvest, including an understanding of stakeholder objectives, incentives and behaviour, (ii) a sound understanding of the biological system and (iii) responses to intrinsic (density dependent) and extrinsic (environment, harvest) factors. The sustainable harvest also depends on management objectives that are congruent with the renewal capacity of the resource. Even with a firm commitment to long-term resource conservation, harvest managers will always be burdened by complex, dynamic systems that are only partially observable, and by management controls that are indirect and limited.

Decision-making Framework for Harvest Management

The move toward accountability and explicitness in natural resource management has led to a need for a more structured approach to decision-making. Improved clarity about key elements in a decision-making process can help decision-makers focus attention on what, why, and how actions will be taken, as well as the likely impacts of these actions. Furthermore, consideration must be given to stakeholders involved in the decision-making or the implementation of policy: who has the authority, responsibility and resources to implement actions. It is important to acknowledge distinctions between those designated as decision-makers and other stakeholders who may implement actions or be affected by them, as well as their respective roles and influence in the decision-making process.

Gaining knowledge and information is vital throughout the decision-making process, not only biological population or ecological data but also social data, i.e. information about human usage of a resource, user goals, motivations and incentives, and the interactions amongst different user groups and institutional organizations. Following a structured decision-making process helps frame management decisions and tasks in the broader socio-ecological context, whereby engagement with stakeholders, the formulation of management options, the sharing of knowledge and information, a greater understanding of uncertainties and acknowledgement of risk can lead to better management decisions and their effective implementation.

Activities in a structured approach to decision-making include the following:

- Developing a shared understanding of the problem in a socio-ecological context
- Setting up an appropriate organizational structure for the planning process
- Engaging the relevant stakeholders in the decision-making process
- Specifying objectives and trade-offs that capture the values of stakeholders
- Setting targets linked to key issues that reflect desired management objectives and outcomes
- Identifying the range of decision alternatives from which actions are to be selected
- Specifying assumptions about resource structures and functions
- Projecting the consequences of alternative actions
- Identifying key uncertainties
- Measuring risk tolerance for potential consequences of decisions
- Accounting for future impacts of present decisions
- Accounting for legal requirements and constraints

Once a good understanding of the social-ecological context is developed, the remaining activities need not be done sequentially, as long as they are done as part of an iterative and learning process. It is this iterative process, whereby decision-makers actively engage, act and learn with stakeholders that can lead to the institutional changes required for ensuring the sustainable population management and harvesting of waterbird populations. Scientists assist the process by providing predictive model tools to better understand the biological system and predict outcomes of management actions as well as providing guidance on the design of monitoring protocols needed to assess the effects of alternative management actions. Scientists can also facilitate the iterative learning process as 'honest brokers' to ensure that the scientific models and evidence are correctly interpreted, and then incorporated in decision-making according to the objectives and potential management actions.

Sustainable management and harvesting of waterbird populations as part of a socio-ecological system

Management of migratory waterbirds is largely about managing human organizations and requires coordination between countries and people who often have different socio-economic, political and cultural settings and values. This is often the case within the EU. Therefore, assessing and taking into account how human interests relate to the status and ecology of the target species is an important prerequisite if international management plans are to be effective and resilient to change in an integrated social and ecological system.

A good understanding of the socio-ecological system is therefore the first step in the process. This requires assessment of:

- The biological/ecological system
- The spatial and temporal scales at which management is to be applied
- The socio-economic-political systems in the geographic range of the target species:
    - Regulatory governance frameworks (international directives, treaties, conventions, Common Agricultural Policy (CAP), national legislation that pursue explicit broad goal-oriented objectives e.g. biodiversity protection and sustainable use)
    - Regulatory regimes (which may encompass a range of cultural aspects: customs, norms, as well as economic market forces that can influence the outcomes of a governance system)
- Stakeholders: who are they and what influence do they have on the system and the target species?

Successful development and implementation of a plan requires understanding the linkages between governance levels (from international to local, and between stakeholder groups) and feedbacks among different elements of the socio-ecological system, from international to local levels. This facilitates dialogue, learning and decision-making amongst institutions and groups, as well as helping connect both top-down (e.g. regulatory instruments) and bottom-up (e.g. co-management of waterbird habitats and hunting organisation via local voluntary agreements between stakeholder groups) initiatives. It is also important to recognize the boundaries of the management plan, i.e. what is realistically inside and outside the scope and influence of the plan in question? For example, a management plan is unlikely to change an international directive or convention, but can influence national/regional regulations or policies.

Organizational Structure

It is important to build sufficient institutional capacity and networks that facilitate decision-making processes and ensure the transfer of knowledge at multiple levels. Building trust between stakeholders is a key to success, and this is best achieved by: (i) an open and transparent process; (ii) communication across all governance levels; (iii) respect for various viewpoints and trade-offs (such as scientific and local knowledge) and (iv) understanding of the participants' roles in the process. Developing and implementing a plan requires leadership and long-term commitment by institutions and by persons who are able to bridge the viewpoints of various stakeholders and have the capacity to keep the process going and secure the funding for its operation.

An organizational structure should reflect the relevant scale, levels and types of governance and stakeholder interests. Organizations that can play an intermediary function between different levels and scales, known as boundary or bridging organizations, can enable the co-production of knowledge and facilitate stakeholder participation with decision-making processes (Cash et al. 2006). Some AEWA Working Groups are an example of such an organization.

A management plan will most often provide a set of alternatives of future scenarios and advice and guidance as part of the decision-making process, while the responsibility for making the ultimate decision will lie with the national regulatory agencies or governments. The mandate of the management plan working group and role of participants should be clearly defined from the outset to reflect its scope.

Information Management

One of the critical aspects of a structured decision-making process is the utilization and application of information (information management). This is particularly the case in relation to the explicit recognition and handling of uncertainties that are inherent within complex socio-ecological systems (Rauschmayer et al. 2008, Berghöfer et al. 2008). Not only are there biological uncertainties but also uncertainties about human aspects of the socio-ecological system, e.g. behavioural responses to and economic consequences of management actions.

A first step is to recognise and clarify potential sources of information, not only scientific but also local experience-based knowledge and expertise (Berghöfer et al. 2008). Hunters can be a good source of information for scientists, providing insights about the local dynamics of waterbird populations, their behaviour and resource use. In addition, understanding the goals, motivations and behaviours of hunters can help gauge the likely impact of management actions and ultimately their effectiveness. In order to gain a fuller understanding of the socio-ecological system requires collaboration between ecological and social scientists. Information requirements will also depend upon, and should be aligned with, the likely management strategies to be employed. The challenge then is to integrate and transform these different knowledge types into formats that are accessible and usable for those involved in the decision-making process. Locally derived experience-based knowledge is often generated over shorter time spans than scientific knowledge. This has additional implications for integrating both knowledge types into an iterative decision-making process where knowledge availability and its timing may not coincide with key decision points. The synthesis of information and knowledge is likely to require new formats, mechanisms and a willingness to share it, not only between scientists, managers and lay stakeholders but also between scientific disciplines e.g. the biological and social sciences (Berghöfer et al. 2008).

The use of information within adaptive management, as part of an iterative decision-making process, can help tackle issues related to uncertainty. The inclusion of stakeholders as part of this process, combining scientific and localized knowledge can help develop solutions that are tailored to a specific context and ensure effective implementation of management actions by involving relevant actors. To enable this deliberative and discursive process will require new organizational (institutional) structures and arrangements to engender and sustain collaborations for the adaptive co-management of waterbird populations. These organizational structures will, in turn, need to be able to adapt and change as the context of the situation and socio-economic system changes.

Adaptive Management[2]

Adaptive management is itself a structured approach to decision-making, in that it includes the key elements described above. The distinguishing features of adaptive management are its emphasis on sequential decision-making in the face of uncertainty and the opportunity for improved management as learning about system processes accumulates over time. Adaptive management can be described in terms of a set-up or planning phase during which some essential elements are put in place, and an iterative phase in which the elements are linked together in a sequential decision-making process (Fig. 1). The iterative phase uses the elements of the set-up phase in

[2] Portions of this text excerpted with permission from Williams et al. (2007).

an on-going cycle of learning about system structure and function, and managing based on what is learned.

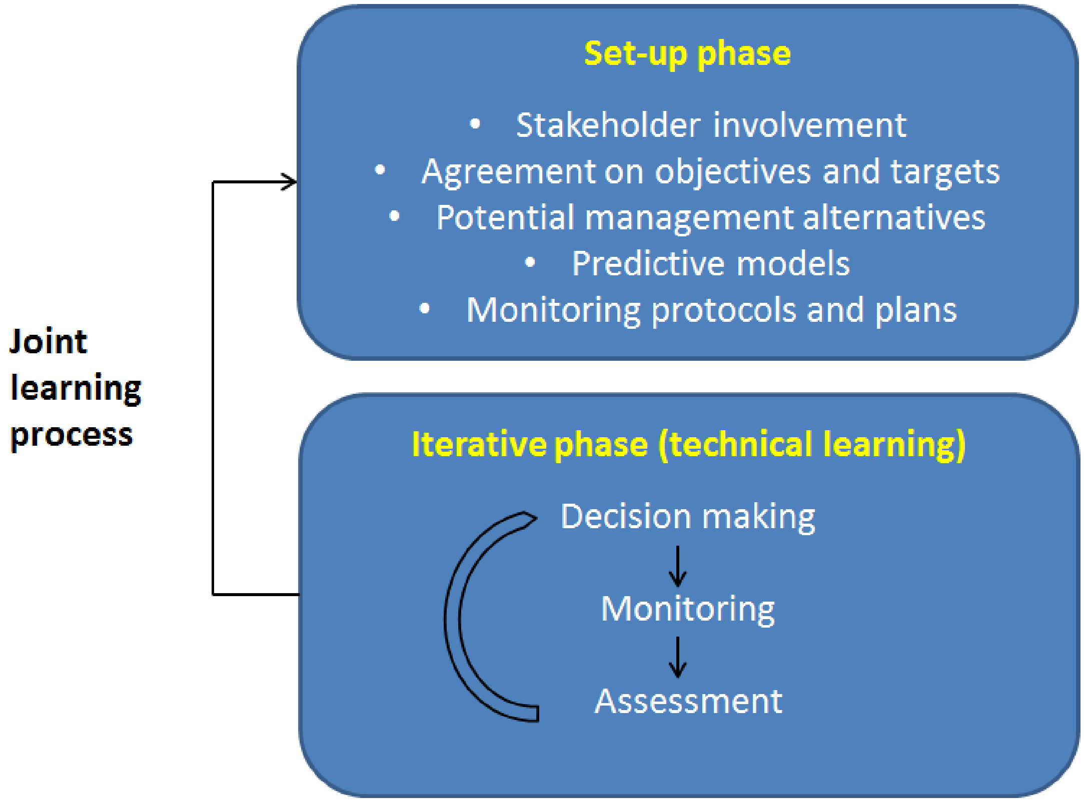

Fig. 1. Process of adaptive management (from Williams et al. 2007).

The elements in the set-up phase of adaptive management include: (i) stakeholder involvement, (ii) objectives, (iii) management alternatives, (iv) predictive models, and (v) monitoring protocols.

Stakeholder involvement. Stakeholders bring different perspectives, preferences, and values to decision-making. It is important to have at least some stakeholder engagement in all the set-up elements of a project, and to continue that engagement throughout the project. A critical challenge is to find common ground that will promote decision-making despite disagreements among stakeholders about what actions to take and why. Failure to engage important stakeholders, and disagreement about how to frame a resource problem and identify its objectives and management alternatives, are common stumbling blocks.

A crucial step to ensure effective stakeholder participation is systematically identifying relevant decision-makers (with the authority or mandate for making decisions), actors (with responsibility for management actions), and other stakeholders (with an interest in the situation but not necessarily authority or responsibility) for consideration within the decision-making process. Stakeholder analysis (*sensu* Conroy and Peterson 2013) can be used to identify and assess the importance of different people, groups or organizations in terms of:

- The ability of the decision to affect the stakeholder
- The stakeholder's ability to affect the decision by asking a series of questions:
    1. Who is potentially affected by the decisions made?
    2. Who is usually involved in similar decisions and (just as importantly) who is normally excluded and why? Should they be included now?
    3. Who can provide knowledge of how the system works, e.g. biologists, ecologists, and social scientists?
    4. Who has the legal authority and is able (i.e. has the resources) to implement management actions?

Distinguishing between decision-makers, actors and other stakeholders can help clarify the role of these different stakeholder types in the decision process and their influence on outcomes. Once initial stakeholders have been identified it must be borne in mind that these stakeholders can (are likely to) change over time or as new situations arise. A flexible approach for stakeholder participation is likely to be needed, using a variety of participatory methods, to encourage different stakeholder groups to learn from each other over time.

Objectives. Successful implementation of adaptive management depends on a clear statement of project objectives. Objectives represent benchmarks against which to compare the potential effects of different management actions, and serve as measures to evaluate the effectiveness of management strategies. Objectives can often be represented by measurable targets, but these must be explicitly linked to the issues under consideration and desired management outcomes. Targets, such as those for populations and habitats, should be informed by scientific assessments but must be agreed upon as part of the collaborative decision-making process, as they are socially constructed measures.

Management alternatives. Adaptive decision-making requires the clear identification of a set of potential alternatives and scenarios from which to select an action at each decision point. Some actions might affect the resource directly; others might have indirect effects. Learning and decision-making both depend on our ability to recognize differences in the consequences of different actions, which in turn offers the possibility of comparing and contrasting them in order to choose the best action.

Predictive models. Models play a critical role in adaptive management, as expressions of our understanding of the resource, as engines of ecological inference, and as indicators of the benefits, costs, and consequences of alternative management strategies. Importantly, they can represent uncertainty (or disagreement) about the resource system. Models are used to characterize resource changes over time, as the resource responds to fluctuating environmental conditions and management actions. Where data allow, predictive models may be complex, but in situations where data are more fragmentary models may simply be conceptual and reflecting professional judgment.

Monitoring protocols. Monitoring provides the information needed for both learning and evaluation of management effectiveness. The value of monitoring in adaptive management is inherited from its contribution to decision making. To make monitoring useful, choices of what ecological and socio-economic attributes to monitor and how to monitor them (frequency, extent, intensity, etc.), must be linked closely to the management situation, objectives and targets that motivate the monitoring in the first place, as well as practical limits on staff and funding. While monitoring the ecological sustainability has been an integral part of the development of the adaptive management approach, monitoring the effect of decision making on social and economic sustainability is also an important part of the process to ensure that decisions can be successfully implemented.

In the <u>iterative phase</u> of adaptive management, the elements in the set-up phase are folded into a recursive process of decision making, follow-up monitoring, assessment, learning and feedback, and institutional learning.

Decision-making. The actual process of adaptive decision-making entails decisions at recurring points in time that reflect the current level of understanding and take into account future scenarios and consequences of decisions. Decision-making at each decision point considers management objectives, resource status, and knowledge about consequences of potential actions. Decisions are then implemented by means of management actions on the ground.

Follow-up monitoring. Monitoring provides information to estimate resource status, underpin decision-making, and facilitate evaluation and learning after decisions are made. Monitoring is an on-going activity, conducted according to the protocols developed in the set-up phase.

Assessment. The data produced by monitoring are used along with other information to evaluate management effectiveness, understand resource status, and reduce uncertainty about management effects. Learning is promoted by comparing predictions generated by the models with data-based estimates of actual responses. Monitoring data can also be compared with targets representing desired outcomes, in order to evaluate the effectiveness of management and measure its success in attaining management objectives.

Learning and feedback. The understanding gained from monitoring and assessment helps in selecting future management actions. The iterative cycle of decision-making, monitoring, and assessment, repeated over the course of a project, leads gradually to a better understanding of resource dynamics and an adjusted management strategy based on what is learned.

Institutional learning. Periodically it is useful to interrupt the technical cycle of decision-making, monitoring, assessment, and feedback in order to reconsider project objectives, targets, management alternatives, trade-offs and other elements of the set-up phase. This may be necessary because the socio-ecological system changes in a direction that was not originally foreseen and it may require a change in the stakeholders involved in the process. This reconsideration constitutes an institutional learning cycle that complements, but differs from, the cycle of technical learning. In combination, the two cycles are referred to as "double-loop" learning.

U.S. Waterfowl Management and Applicability to Europe

Contribution lead by Fred A. Johnson, Southeast Ecological Science Center, U.S. Geological Survey, Gainesville, Florida, USA

Waterfowl management in Europe faces similar challenges in relation to the management of migratory waterbirds as in North America, i.e. waterbird populations migrate through a number of countries between their breeding and wintering areas during their annual cycles and they are subject to various management actions including harvest. While the U.S. has worked to establish coordinated harvest management across flyways, a similar mechanism has not been established across the African-Eurasian flyway. Countries within the European Union (EU) have management plans but consideration of cumulative impacts across the flyway has not been formalized. The following section will review the experience with coordinating waterbird management in the U.S. and will assess its applicability in Europe.

The U.S. government's authority for establishing waterfowl hunting regulations is derived from treaties for the protection of migratory birds signed with Great Britain (for Canada in 1916), Mexico (1936), Japan (1972), and the Soviet Union (1978) (U.S. Fish and Wildlife Service 1975). These treaties prohibit all take of migratory birds from March 10 to September 1 each year, and provide for hunting seasons not to exceed 3½ months. Each year, the U.S. Fish and Wildlife Service (USFWS) solicits proposals for hunting seasons from interested parties, and after extensive public deliberations, establishes guidelines within which States select their own hunting seasons. States may be more restrictive, but not more liberal, than federal guidelines allow. Hunting regulations typically specify season dates, daily bag limits, shooting hours, and legal methods of take. Optimal sustainable harvest for a given bird population could similarly be computed at the scale of the flyway population, then split into maximum national harvests, with Member States being free to implement a more restrictive total bag for their own national hunters.

Waterfowl hunting regulations have worked reasonably well in North America, as evidenced by levels of hunting opportunity and harvest that have been maintained for at least 30 years. This record of success is notable, given that natural resources often are over-exploited to the point of economic extinction (Ludwig et al. 1993). This is not to say, however, that the process of setting waterfowl hunting regulations has been without problems. The process is often plagued by controversy, contentiousness, and, on occasion, court challenges and Congressional intervention (Feierabend 1984, Babcock and Sparrowe 1989, Sparrowe and Babcock 1989). These difficulties stem from uncertainty (or disagreement) about the impacts of regulations on harvest and waterfowl abundance, and from harvest management objectives that are often vague, ambiguous, or incommensurate (Johnson et al. 1993). In the face of these ambiguities, the USFWS traditionally took a conservative approach to hunting regulations, thereby exacerbating the potential for conflict, particularly during periodic downturns in waterfowl abundance (Blohm 1989).

Beginning in the mid-1980s, the USFWS began searching for ways to improve the regulation of waterfowl harvests. An effort to stabilize regulations, and thus avoid much of the annual debate about appropriate regulatory responses to environmental variation, was eventually abandoned (U.S. Fish and Wildlife Service 1988). The search for an alternative approach intensified in the 1990s, when large changes in the abundance of ducks prompted renewed controversy about appropriate harvest levels. Eventually, improvements in the regulatory process were framed in terms of adaptive resource management, in which there is an explicit accounting for uncertainty as to management impacts, and for the influence of management actions on reducing that uncertainty (Williams and Johnson 1995). Since 1995, mallard hunting regulations in the United States have been prescribed by a formal process referred to as adaptive harvest management (Johnson et al. 1996). More recent efforts have extended the process to include other species of migratory game birds.

Many European waterfowl managers and ecologists have called for an adaptive approach to the management of European duck harvest (e.g. Elmberg et al. 2006). Despite such recommendations, managers and stakeholders may misunderstand the process of adaptive management to the extent that they believe the approach could not be usefully implemented in Europe. Here, we consider possible concerns.

We recognize that North American waterfowl management involves three countries and two languages, whereas European waterfowl management involves a large number of countries, languages, and hunting traditions. This variation within Europe, however, does not preclude agreement on a framework for management. If agreement on objectives and management actions is possible, then an adaptive approach to management can still be taken within each country or group of countries for which agreement is possible. In such a case, the actions of non-participating countries would be basically viewed as components of environmental variation, in the sense of uncontrollable and possibly even unpredictable effects on populations. The central point is that any country or group of countries claiming interest in management could choose to manage and follow an adaptive approach in doing so. In the European context, the EU Birds Directive presents a common management framework for the EU Member States and each one of them are bound by the requirements of the Directive including the one that they shall ensure that their hunting of the species listed in Annex II does not jeopardise conservation efforts in their distribution area. In the broader flyway context, the AEWA requirements of (i) insurance of sustainable use of populations listed in Column B and (ii) adaptive harvest management of populations listed in Column A with the objective of restoring them to favourable conservation status call for adaptive approaches.

Other concerns may emphasize variation among European countries in hunting regulations and systems, and in magnitudes of harvest. Such concerns may not recognize the geographic variation that exists within North America in regulation packages (these vary among the North American flyways), in timing of open seasons, and in magnitudes of harvest. Adaptive harvest management does not require geographic uniformity in regulations nor harvest, but instead represents an extremely flexible approach that can readily accommodate geographic variation. Actually, if an optimal sustainable absolute harvest can be derived for a population at the flyway scale, this could then be split into maximum national harvests, with countries later being free to recommend/implement a more restrictive total bag for their own hunters.

Additional concerns may be that European waterfowl monitoring programs are not nearly as well developed as those established in North America, and that much less is known about populations in Europe than in North America. While these differences may be true, they do not argue against the use of adaptive management. Some sort of monitoring is indeed necessary for adaptive management, but precise estimates of population size and all demographic rate parameters are not a requirement. This document will demonstrate that adaptive management can be practiced with minimum information requirements and many of them are already in place in Europe. Similarly, because of its emphasis on learning, adaptive management is actually more important for poorly known systems than for those that are well understood. Adaptive management simply provides a means of managing in a manner that is optimal with respect to knowledge about the managed system.

While we recognize that concerns have been expressed regarding adaptive approaches to population management in Europe, these concerns can be addressed and the benefits of this approach can be realized. The AEWA adaptive flyway management plan for the Svalbard pink-footed goose has provided some first positive results and proof of concept with regard to establishing stakeholder consensus about objectives and actions as well as generating learning (see http//:pinkfootedgoose/aewa/info and Case Study 1).

Case Study 1: AEWA International Species Management Plan for the Svalbard population of pink-footed geese – the first European test case of adaptive flyway management

Jesper Madsen

Having recovered from low population sizes during the last century many populations of wild geese in Western Europe are flourishing following a combination of improved protection, land use changes that create food for geese, and climate change. Because geese often congregate to forage on farmland which has become more intensively cultivated, conflicts between goose conservation and agricultural interests have become common. Solutions to conflicts have differed widely among countries depending on the political willingness to pay economic compensation to farmers. However, even in countries where compensation or subsidies are provided to farmers conflicts tend to continue because the goose populations are still increasing. Hence, in several countries farmers ask for more economic support, but politicians are reluctant to provide aid because of budget cuts and fear of creating entitlements. Consequently, farmers in some countries have requested that populations be controlled to stop agricultural losses. This has been suggested for breeding geese in Scotland and the Netherlands; as well with migratory pink-footed geese in Norway. With regard to the breeding stocks, the political decision to cull populations lies with the national governments; however, with migratory species, the issue is international and there is little European history of trans-boundary coordination of wildlife management.

The Strategic Plan for 2009-2017 of AEWA recognizes that international coordination and flexible instruments are needed to manage waterbird harvest. The Svalbard pink-footed goose has been selected as the first test case for an AEWA International Species Management Plan for the following reasons. Firstly, conflicts have escalated due to increasing goose numbers and spring-staging geese feeding on pasture grass and newly-sown crops (Tombre et al. 2013). Secondly, these geese have degraded tundra habitats in Svalbard and this damage might be increasing (Petersen et al. 2013). Thirdly, Norwegian farmers and management authorities had agreed to set a population target of about 40,000 without consulting other range states (Denmark, The Netherlands and Belgium). Therefore, an internationally coordinated adaptive management process was initiated that included nature agencies from the four range states, organisations (representing hunters, BirdLife and farmers) and scientists. This international working group reached consensus on problems, objectives and alternative actions to 'maintain a favourable conservation status for Svalbard pink-footed geese while taking into account economic and recreational interests' (Madsen and Williams 2012). This plan was endorsed by the Meeting of the Parties to AEWA in 2012, after in depth consultation among affected parties, and implemented by the range states. One of the most controversial objectives, and novel to European waterbird management, was to stabilise the population at 60,000 individuals, using hunting as the tool. The species is huntable species in Norway and Denmark, but it is protected in The Netherlands and Belgium. The Netherlands and Belgium do not want a hunting season; hence, it is between Denmark and Norway to achieve the objective. The agreed population target is a social construct (Williams and Madsen 2013). Preliminary model evaluations suggested that a lower target could be justified from a population viability perspective, provided that hunting could be effectively regulated, so Norwegian representatives argued for a target below 60,000. Oppositely, BirdLife representatives argued that in principle, populations should be allowed to fluctuate naturally, but also recognized that continued increase of the goose population would potentially cause loss of biodiversity in Svalbard and lead to more conflict; hence they accepted the stabilisation target. An important reason the parties agreed to the target and the tool was the implementation of an adaptive management framework to predict effects of harvest on population size, monitor, evaluate and revise the harvest of geese (Johnson et al. 2014). Hence, scientific credibility and transparency were important. Furthermore, Denmark and Norway have agreed to make an emergency closure of the hunting season if the population is predicted to decline below the target. The coming years will show if hunting regulations will work. If they do not, the working group will discuss alternative actions.

Adaptive management has provided the structure to deal with these politically delicate issues and, at the same time, reduce uncertainties in the understanding of the population dynamics and responses to changes in harvest regimes. Building trust between parties and having a joint learning process have been important cornerstones in the process. Hunters have expressed a desire to be seen as partners contributing to management of the system. Science-led projects have been established in Norway and Denmark to explore how voluntary agreements with hunting organisation can be used as co-management tools to regulate the harvest. In the coming years there is room for greater harvests, to the benefit of hunters, and hunting regulations in Denmark have been liberalised accordingly with an extension of the hunting season length. Interestingly, hunters know that if adaptive management is successful, they will have to reduce their harvest in order to meet population objectives, which demonstrates commitment to joint stakeholder desires and the management process.

Information Requirements

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The following section reviews the minimum information requirements necessary to set-up a sustainable harvest-management regime. In some cases, more detailed demographic information may be available and permit a more rigorous assessment.

In the absence of (or ignorance about) density dependence, one can use the realized growth rate of the population to determine the harvest required to meet some goal for population growth. For a stable or growing population (N) with observed growth rate $\lambda_{obs} \geq 1$, the harvest rate (h_t) and absolute harvest (H_t) needed to achieve a desired growth rate $\lambda_{obj} \leq \lambda_{obs}$ are:

$h_t = 1 - \frac{\lambda_{obj}}{\lambda_{obs}}$ and $H_t = N_t h_t$.

To stabilize a growing population, the harvest rate needed is:

$h = \frac{\lambda_{obs} - 1}{\lambda_{obs}}$.

For a declining population with observed $\lambda_{obs} < 1$, the harvest rate needed to achieve a desired growth rate λ_{obj}, let:

$\lambda_{obs} = \lambda_{noHunt}(1 - h_{obs})$.

If h_{obs} can be estimated, solve for λ_{noHunt}, set $\lambda_{obs} = \lambda_{obj}$ and solve for h_{obj}. If h_{obs} cannot be estimated, an estimate of λ_{max} can be substituted for λ_{noHunt} if population size is believed to be very low relative to carrying capacity, K (i.e. h_{obj} will be an upper bound).

To account for density dependence, one of the most simple and commonly used models to determine sustainable harvests for birds is the theta-logistic:

$N_{t+1} = N_t + N_t r_{max} \left[1 - \left(\frac{N_t}{K}\right)^{\theta}\right] - h_t N_t$,

where r_{max} is the intrinsic rate of growth. The theta-logistic lacks age or sex structure and thus should be considered a first approximation if reproductive or survival rates are likely to be strongly stage specific, which is often the case in waterbirds. The harvest rate and harvest for the maximum sustainable yield (MSY) are (Johnson et al. 2012):

$h_{MSY} = r_{max}\frac{\theta}{(\theta+1)}$ and

$H_{MSY} = r_{max}K\frac{\theta}{(\theta+1)^{\frac{(\theta+1)}{\theta}}}$,

and the population size associated with MSY is (often referred to as the level of maximum net productivity):

$N_{MSY} = K(\theta+1)^{\frac{-1}{\theta}}$.

For the standard logistic with linear density dependence ($\theta = 1$), the management parameters simplify to:

$h_{MSY} = \frac{r_{max}}{2}$, $H_{MSY} = h_{MSY}\frac{K}{2}$, and $N_{MSY} = \frac{K}{2}$.

The dangers of harvesting at a constant level H_{MSY} are well known (Ludwig 2001), but harvest is more likely to be sustainable if a constant harvest rate is used instead (i.e. absolute harvest is changed to reflect stochastic changes in population size) (Runge et al. 2004):

$H_t = h_{MSY}N_t$.

Thus, the best practice would be to assess constant optimal harvest *rate*, and from the regular monitoring of population abundance calculate the allowable harvest. This approach also may be relatively easy to implement in the field; e.g. through the establishment of harvest quotas, season lengths and/or bag limits.

Long periods between estimates of population size, however, increase the risk of over-harvest (Johnson et al. 2012). Total waterfowl population sizes are regularly and relatively precisely monitored in the Western Palearctic, with long-term International Waterbird Censuses in place throughout the region since the mid-1960s. However, the fact that such counts are carried out in so many different countries, hence implying a long list of national coordinators and a variety of constraints, means that total population size estimates can only be produced episodically and with a few years delay after the censuses themselves. For such reasons a coherent framework for managing ecological risk is necessary.

To account for management objectives other than MSY, Runge et al. (2009) and Johnson et al. (2012) proposed the inclusion of a parameter F to account for a population objective other than N_{MSY}, such that the allowable harvest is:

$H_t = r_{max}\frac{\theta}{(\theta+1)}FN_t$, where $0 \leq \mathrm{F} < \frac{(\theta+1)}{\theta}$ (for $\mathrm{F} = 1$, the management objective is MSY). If a goal for population size can be expressed as a fraction of K, the appropriate value of F can be determined by solving:

$$\frac{N}{K} = \left(1 - F\frac{\theta}{(\theta+1)}\right)^{\frac{1}{\theta}}.$$

If the goal for population size is expressed as an absolute value, then an estimate of K must be available. Unfortunately, K can be difficult to estimate without a long time series of population and harvest estimates.

We can also use this approach to ask about status of the current harvest by working backwards through the above formulations. The relevant questions are whether the harvest is sustainable and, importantly, what management objective and attitude toward risk are reflected in the observed harvest. In other words, is the harvest sustainable biologically and, if so, does it accurately reflect the decision-makers' objectives and attitude toward risk?

Finally, we note that additional complexity can be included in this approach in a straightforward manner. For example, significant environmental variation can be included in the theta-logistic model (Johnson et al. 2012). In this case, it might be more appropriate to use dynamic optimization methods to calculate a state-dependent strategy (Williams 1985) rather than a constant harvest rate. If there is pronounced stage structure in a waterbird population, matrix models (Hauser et al. 2006) and their logistic analog (Jensen 1996) might be more appropriate than the unstructured logistic model described here.

So how does one go about estimating the parameters needed to apply this approach? The observed growth rate λ_{obs} can be estimated from a series of population estimates. Let:

$$N_t = N_0 e^{rt},$$

and then use simple linear regression of population size against time:

$$ln(N_t) = \beta_0 + \beta_1(t) + \varepsilon,$$

so that the time-averaged $\lambda_{obs} = e^{(\beta_1+\varepsilon)}$, where $\varepsilon \sim Normal(0, \sigma^2)$. If estimates of the variance of population estimates are not available, then ε represents both observation and process error; if variance estimates are available, then a Bayesian hierarchical model can be used to get an estimate of process error alone (i.e. a state-space model). The growth rate λ_{obs} can also be estimated from empirical estimates of reproduction and survival if available (e.g. using a matrix population model, Caswell 2001).

A variety of approaches are available for estimating the parameters of the theta-logistic model, recognizing that any approach must estimate r_{max} for rarified populations under ideal conditions (Johnson et al. 2012). Perhaps the approach requiring the least amount of information is from Niel and Lebreton (2005):

$$r_{max} = \left[\frac{(sa-s+a+1)\sqrt{(s-sa-a-1)^2-4sa^2}}{2a}\right] - 1,$$

where a is age at first breeding and s is the survival rate of animals of reproductive age. Age at first breeding is often known, but how can survival be estimated under ideal conditions? Johnson et al. (2012) proposed a model based on detailed mortality records from 1,111 captive individuals of 23 bird species ranging in mass from 12 to 8,663 grams:

$$s = p^{1/(exp(3.22+0.24log(M)+e)-a)}$$

where M is body mass (kg), p is the estimated proportion of the population remaining alive at the maximum observed life span and:

$p \sim Beta(3.34, 101.24)$ so that $p_{mean} = 0.03(sd = 0.017)$,

and $\varepsilon \sim Normal(0, \sigma^2 = 0.087)$.

If more detailed demographic information is available, one can use the method of Slade et al. (1998) to estimate the intrinsic growth rate:

$$1 = s_a\lambda^{-1} + s_j f\lambda^{-\alpha} - s_j f {s_a}^{(\omega-\alpha+1)}\lambda^{-(\omega+1)},$$

where λ is the maximum finite population growth rate (i.e. $r_{max} = \lambda - 1$), s_j and s_a are fixed survival rates of pre-reproductives and adults, respectively, f is a fixed fecundity for all reproductives, and α and ω are ages at first and last breeding, respectively. If $a = 1$ and there is no reproductive senescence (Holmes et al. 2003), this can be simplified to $\lambda = s_a + s_j f$. Johnson et al. (2012) provide methods for estimating f and s_j for birds.

To assess the uncertainty of sustainable harvests, we can account for the stochastic nature of the components. First, one can generate a large number of random deviates of mass M and proportion of the population remaining at the observed maximum lifespan p using Gamma and Beta distributions, respectively, based on the means and standard deviations provided. These deviates can be used with the known or assumed age at first breeding to generate a large number of deviates of survival s. Then one can use the deviates of s and the age at first breeding to generate a large number of random deviates of r_{max} using the formula by Niel and Lebreton (2005). Next, one can use the deviates of r_{max} to generate a large number of deviates of θ using the model provided by Johnson et al. (2012):

$$ln(\theta) = 1.129 - 1.824r_{max} + \varepsilon,$$

where $\varepsilon \sim Normal(0, \sigma^2 = 0.942)$. Alternatively, one could assume $\theta = 1$, which would generally be a conservative approach for waterbirds. Then generate a large number of random deviates of the management parameter F for values of the population objective(s). Finally, generate deviates of population size N from the available sampling distribution. The frequency distribution of the harvest rate h or the harvest H can then be simulated using the random deviates and equations provided. Assuming that decision-makers can express their risk tolerance on a 0 – 1 scale (with 0 being completely risk averse), the level of risk tolerance (R) can be used to select a quantile from the simulated distribution. We suspect that most managers will exhibit risk-averse or risk-neutral attitudes ($0 < R \leq 0.5$) rather than risk-seeking behaviour ($0.5 < R \leq 1$).

In conclusion, an informed approach to setting allowable harvests does not require detailed demographic information. Essential to the process, however, are estimates of either the observed growth rate from a monitoring program or the growth rate expected under ideal conditions. The latter can in turn be based on empirical data or on the allometric model described above. In addition, periodic estimates of population size are needed, as well as either empirical information or reasonable assumptions about the form of density dependence. We stress that whatever the source of information, managers should strive to account for uncertainty in demographic parameters, as well as for the decision-makers' objectives and attitude toward risk. In the case studies 2 and 3 we provide examples of how levels of sustainable harvest have been estimated.

Case Study 2: Harvest management of a rare game waterbird: the red-crested pochard in Camargue (France) and Europe

Pierre Defos du Rau & Jean-Yves Mondain-Monval

The Central Europe and Western Mediterranean population of Red-Crested Pochard (RCP) *Netta rufina* is estimated at 50,000 individuals (Wetlands International 2014). The total annual harvest for this population was estimated at 8,000 birds in the mid-1980's (Shedden 1986), 700 of which being harvested in the Camargue (France) on average, and the rest in Spain. To investigate sustainability of RCP harvest, we examined the relationship between harvest levels and RCP population dynamics through assessment of theoretical maximum sustainable harvest level (Niel & Lebreton 2005) at the European scale.

We used the ONCFS long-term duck harvest survey that samples hunting bags in the Camargue over 38,137 ha of hunting estates, encompassing an average 19.6% of the total hunted area. Total RCP harvest for the Camargue was then estimated by area expansion, the sampled estates being evenly distributed both spatially and ecologically around the whole study area. This harvest estimate was added to a 20% crippling loss (Anderson & Burnham 1976). An index of harvest rate for the Camargue RCP population was computed as the ratio of the estimated total harvest of the corresponding year to an estimate of the total number of birds available for harvest (i.e. the total number of birds passing through the harvest area during the hunting season), estimated as twice the maximum number counted from aerial surveys over the Camargue (Caizergues et al. 2011, Gourlay-Larour et al. 2013). Using this method, from 1988 to 2005 the estimated harvest rate of RCP in the Camargue averaged 17.4%, with strong inter-annual variations (SD = 13.8%).

Because of the scarcity of data for many vital rates of RCP, we used the demographic invariant method (Niel and Lebreton 2005) to evaluate the impact of harvest on RCP population growth rate (λ_{max}) through comparison of the potential excess growth to the estimated total number of harvested RCP.

The potential maximum harvestable population fraction allowed by excess growth was estimated following Wade (1998) as:

$$P = N\beta(\lambda_{\max} - 1)$$

where N is the total population size, currently estimated at 50,000 individuals (Wetlands International 2014) and *b* is a correction factor accounting for the effect of density on demographic performance and set at the default value of 0.5 as recommended by Wade (1998) and Niel and Lebreton (2005) in the absence of further information on any species-specific density-dependence process. We estimated λ_{max} following Niel and Lebreton (2005) by solving numerically:

$$\lambda_{max}=\exp([a+So/(\lambda_{max}-So)]^{-1})$$

where So is adult survival probability and a is the average age at first reproduction, both under optimal growth condition. We applied Devineau et al.'s (2006 & in prep.) allometric method to derive So = 0.753 (SE= 0.138) as a function of log(body mass) calculated from the marked RCP sample from Tour du Valat (943g, SD = 105g, n=150). The average age a was estimated at 1.3 years, assuming 70% birds first breed at one year of age (Blums et al. 1996), and the remaining on the next.

The parameter *P* can be interpreted as the maximum number of RCP that can be harvested by any non-natural source of mortality, including hunting and crippling loss but also lead poisoning for instance, without causing decline of the population. However, this estimate of *P* only allows diagnostic of unsustainable harvest, but does not provide confirmation of sustainable exploitation (Niel and Lebreton 2005).

We estimated λ_{max} = 1.54, which in turn yields an estimate of maximum harvestable population fraction *P* = 13,500. From 1988 to 2005, the estimated total harvest of RCP in the Camargue was on average 807 birds, with strong interannual variation (SD = 422). Assuming similar trends in harvest bags in France and in Spain, starting from the last bag evaluation in the mid-80's (Shedden 1986), the total current European RCP bag would be approximately 9,220 birds. This represents 68% of the maximum sustainable harvestable population fraction estimated using the Demographic invariant method. However, this diagnostic is based on several assumptions, including assumed change in harvest bags in Spain, an updated survey of which would be highly desirable. Because some non-natural mortality sources remain unknown, the fact that estimated European harvest levels were well below the predicted maximum sustainable harvest level does not mean that current RCP hunting is sustainable (Niel and Lebreton 2005), all the more as long as up-to-date harvest data are lacking for Europe.

Case Study 3: Taiga bean goose sustainable harvest levels

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We estimated sustainable levels of harvest for the taiga bean goose as part of the ongoing development of an International Single Species Action Plan under the auspices of AEWA. We emphasize that our estimates are a first approximation because detailed demographic information is lacking for taiga bean geese. Our methods are intended to demonstrate how decision-makers can explicitly account for management objectives, uncertainty, and degree of risk tolerance. Using allometric relationships, we estimated parameters of the theta-logistic population model (Johnson et al. 2012). Estimates of the intrinsic rate of growth were $r_{max} = 0.150\ (sd = 0.019)$ and the form of density dependence was $\theta = 2.77\ (sd = 4.72)$, suggesting the strongest density dependence occurs when the population is near carrying capacity. We estimated Potential Take Level in terms of both a constant harvest rate and an absolute harvest from a spring population of 50,000 birds (Fig. 2). We used a management objective to maximize sustainable harvest, although the implications of other management objectives could easily be assessed. We accounted for uncertainty in demographic rates of taiga bean geese, and examined levels of risk tolerance of 0.10, 0.25, and 0.50 on a scale of 0-1 (where 0 is completely risk-averse and 0.5 is risk-neutral; we did not examine risk-seeking behaviour). The allowable harvest of taiga bean geese from a spring population size of 50,000 was less than 5,000 under all scenarios considered. The harvest prior to 2014 (when Finland closed their hunting season) appears to be higher than what we calculated as allowable. This does not necessarily mean, however, that the harvest was unsustainable. It does appear, however, that harvests in excess of 5,000 (from a population of 50,000) represent risk-seeking behaviour, a population objective of less than that required for maximum productivity, or both.

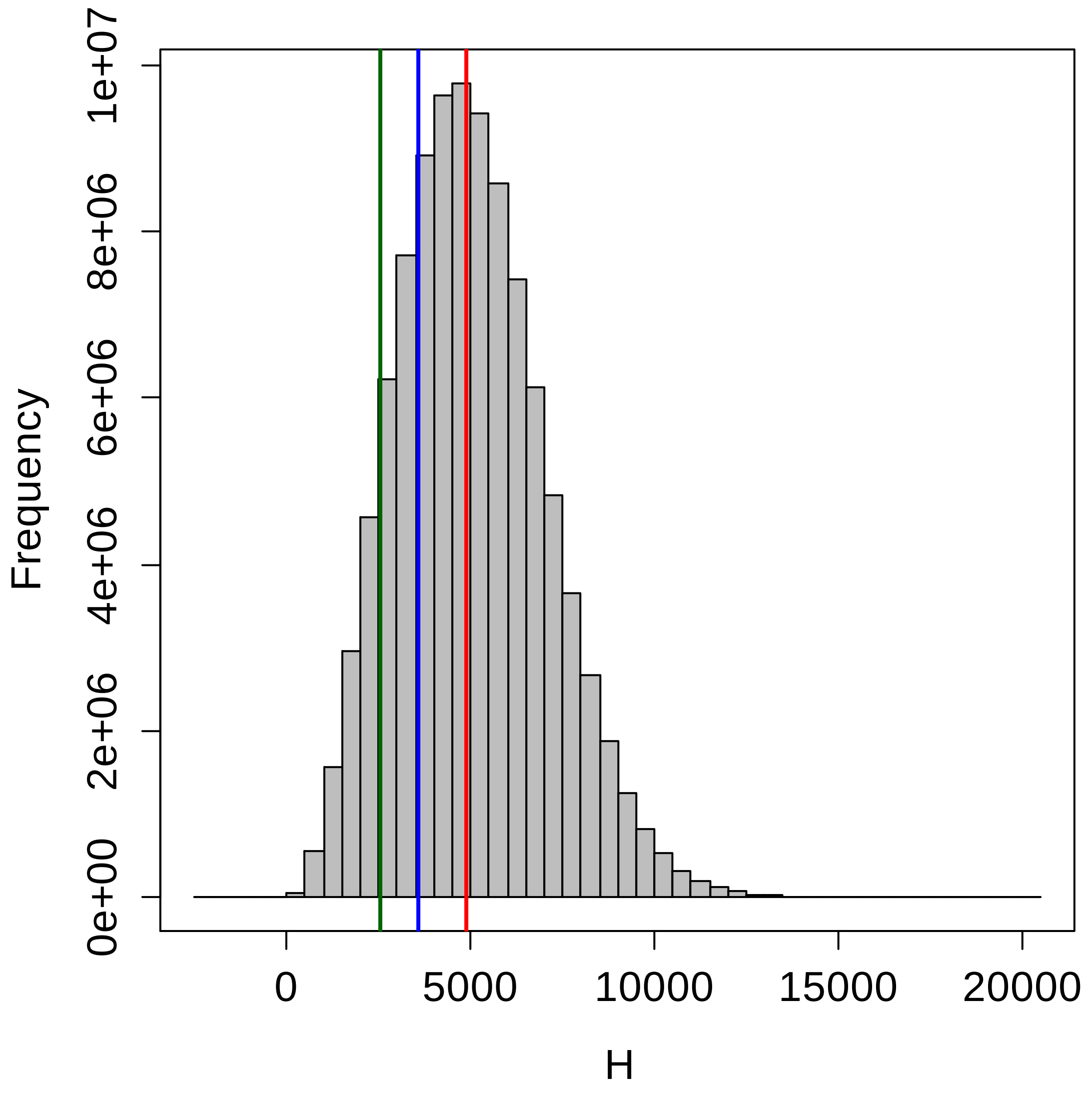

Fig. 2. Distribution of Potential Take Level (PTL) for taiga bean geese harvest, based on a breeding population size of 50,000, model-based values of θ in the theta-logistic model, and an objective to maintain population size at 75% of carrying capacity. The green, blue, and red lines represent PTL for risk-tolerance levels of 0.10, 0.25, and 0.50, respectively.

Availability of Population and Demographic Data for Waterbirds in Europe

As mentioned earlier, European wildlife managers often consider the North American Adaptive Harvest Management schemes as a distant goal almost impossible to reach, because much less is known about populations in Europe than in North America (Nichols et al. 2007). However, the section on information requirements highlights some methods that can be applied to data-poorer situations like in Europe. This section provides an overview of the demographic information available in the Western Palearctic, and demonstrates that most parameters necessary to assess the sustainability of waterfowl harvest with some methods are already available.

Population size estimates are available for most European waterfowl owing to long term International Waterbird Censuses, either in the form of actual estimates derived from the bird counts themselves or, when too much data is lacking in such counts, through expert opinion (Wetlands International 2014). Counting error is usually not quantified in such schemes, but because the counts are repeatedly carried out on the same sites at regular intervals it is possible to calculate the variance around the estimates. From such long-term data series it is possible to compute current population growth rates and error around such estimates (such as used to evaluate conservation status of birds, e.g. in Nagy et al. 2012).

One potential issue however concerns the delineation of the populations being harvested. Some populations are clearly isolated and have straightforward limits, e.g. Svalbard pink-footed goose (Madsen et al. 2014). In many other cases the ranges of convenient management units (e.g. Scott & Rose 1996) widely overlap with a significant degree of exchange of individuals amongst them (e.g. in common teal *Anas crecca*: Guillemain et al. 2005). Other techniques based on genetic markers (e.g. in red-crested pochard: Gay et al. 2004) or stable isotopes in feathers (e.g. in teal, Guillemain et al. 2014) are however available to test for barriers between populations or links between geographic areas, respectively.

Because waterfowl have been extensively studied in the field and are also commonly kept in aviaries as ornamental birds, such traits like age at first reproduction are generally known. European duck species are all able to breed at the age of one year, though a generally unknown (but likely small) proportion of individuals delays breeding until their second year (only 70% breed at one year; Blums et al. 1996). European geese typically start breeding at 2-3 years (see species accounts in Kear 2005).

Survival rates in nature are known from numerous ringing studies of ducks and geese (e.g. review in Bell & Mitchell 1996 for dabbling ducks). However, survival rate given in Niel and Lebreton's (2005) equation is under ideal condition, i.e. without harvest. Such value is often unavailable to researchers, and very different from the estimates that can be derived from field studies: waterfowl life expectancy in captivity is often over 10 or 20 years, while individuals of the same species in nature generally do not live longer than a few years (e.g. 2.24 years in common teal, after Guillemain & Elmberg 2014). Fortunately, survival rate in ideal conditions can be derived from another allometric relationship after Johnson et al. (2012) from aviary measurements of a range of species. This is mostly based on body mass estimates, which are available for all waterfowl.

An alternative method proposed above allows computation of intrinsic population growth rate from adult and juvenile survival rates and fecundity in the population (Slade et al. 1998). Again, the long history of waterfowl ringing and marking in Europe has allowed estimating adult survival rate in most species (e.g. species accounts in Kear 2005). In some goose species the survival rate of juveniles is also fairly known thanks to large-scale ringing programs of unfledged young on their breeding grounds. The situation is not so favourable in ducks, where ringing mostly occurs on the migration stopovers or the wintering grounds, so that first-year birds are only accessible to ringing several weeks or even months after fledging. Their survival rate during this first period of their life is often considered to be similar to that of the adults during the autumn, yet this could be strongly misleading: an indirect analysis based on wing surveys suggested that survival rates of young common teal and wigeon *Anas penelope* during their first 3 months of life was much lower than that of the adults; considering that juvenile survival during the first autumn is only 50% that of the adults at best is likely conservative (Guillemain et al. 2010, 2013). Johnson et al. (2012) also provide methods for estimating juvenile survival for birds.

Fecundity (number of fledged juveniles per female per breeding attempt) is known for most waterfowl, although sometimes from only a handful of case studies (reviews in Cramp & Simmons 1977, Kear 2005, Baldassarre 2014 for North American species also occurring in Europe). When this is not available from case studies Johnson et al. (2012) provide some guidance to derive the value of the parameter allometrically.

Age at first breeding is known in most cases (see above). Ducks in the wild can generally expect only a few years of life, so it is likely that most individuals will breed until their last year. In such cases there is no reproductive senescence and the equation can be simplified (see other sections). Even in geese, which are longer-lived, it is generally considered that older birds will produce more young (e.g. Raveling 1981), so there too senescence does not necessarily need to be taken into account.

To summarize, in most populations of European waterfowl the necessary demographic data are either available from field studies or possible to derive from allometric relationships for at least some kinds of harvest sustainability estimation analyses to be carried out.

Harvest Data

Collection of waterbird hunting bag statistics has a long history in Europe, in some countries dating back to the first half of the 20th century (Lampio 1983) and nowadays, systems exist in the majority of European countries (http://www.artemis-face.eu/). However, the way in which data are collected and the species and geographical resolution and coverage vary greatly between countries, ranging from annual mandatory online reporting schemes organised by the national authorities responsible for wildlife management (Norway) to voluntary schemes at local/regional level organised by hunting clubs or hunters' national organisations (UK). A barrier to reaching the objective of flyway level coordination is the accessibility of bag data. For some countries, annual quality assured reports providing species-specific overviews of harvest can be retrieved with a time-lag of less than one year after the closure of the hunting season (Norway, Denmark) while in other countries, there is no fixed reporting frequency, and only half of EU Member States appear to provide data online or in official documents. In few countries, bag statistics are supplemented by wing or tail collections, which provides species-specific harvest information and useful additional data on the demography of the harvest (age and sex classes) as well as its temporal distribution (Denmark: geese, ducks, waders).

Within the EU, metadata on the availability and accessibility of national bag statistics are reported voluntarily by EU members of The Federation of Associations for Hunting and Conservation (FACE) to the ARTEMIS database (http://www.artemis-face.eu/). However, there is no coordinated collection and compilation of data. Hence, there is no overview of the total harvest across the countries. Outside the EU, bag statistics are available from some countries (Iceland, Norway) in the AEWA region, but in general, the coverage is poor and/or not accessible.

As shown in this report, harvest information is a prerequisite for assessing the sustainability of exploitation. In general, this information is not available and the data are not up-to-date. Few exceptional cases exist, such as the Svalbard pink-footed goose. This population is only harvested in two range states, Denmark and Norway, for which online and annually updated harvest is available. The harvest data from the previous season is part of the annual monitoring program and are used to update the optimal harvest strategy in the adaptive management framework of the International Species Management Plan, which ensures the sustainability of the harvest and the achievement of one of the key targets of the plan to maintain a stable population size. The Svalbard pink-footed goose international management plan and the associated AHM procedure are clearly paving the way for similar schemes in other European waterfowl species.

Summary and Recommendations

This report shows that the EU Birds Directive and the African-Eurasian Waterbird Agreement at the flyway level provide an adequate legal framework for sustainable management of waterbird populations. Nevertheless, despite the existence of an adequate legal framework, almost half of the populations of species listed on Annex II of the Birds Directive have a declining short-term trend and over half of them are listed in Columns A and B of AEWA, i.e. their hunting could either only continue under the framework of an adaptive harvest management plan or their hunting should be regulated with the view of restoring them to a favourable conservation status. Based on international experience, we argue that EU Member States in isolation cannot attain such objectives, and a structured approach to decision-making (such as adaptive management) is needed, supported with adequate organisational structures at flyway scale. We have reviewed the experience with such an approach in North America and assess the applicability of a similar approach in the European context. Following this we have reviewed and prioritised the information requirements of adaptive harvest management. Our main conclusion here is that the information requirements of adaptive harvest management are not onerous and by no means should block the process in the European flyways for most populations.

We also demonstrate that an informed approach to setting allowable harvests does not require detailed demographic information. Essential to the process, however, are estimates of either the observed growth rate from a monitoring program or the growth rate expected under ideal conditions. In addition, periodic estimates of population size are needed, as well as either empirical information or reasonable assumptions about the form of density dependence. We show that such information exists for many populations, but improvements are needed to improve geographic coverage, reliability and timely data availability.

As the observed growth rate plays a central role in harvest management, and population indices can be used to assess the effectiveness of a given set of harvest regulations, it is essential to maintain and further strengthen the monitoring systems that can provide regularly updated population size estimates. Since most of the waterbird populations listed on Annex II are shared with other countries outside of the European Union, it is essential to have data collection and observation networks coordinated between the EU and AEWA (i.e. the EU Birds Directive Art. 12 and the AEWA national reporting processes and the monitoring schemes feeding information into these reporting processes). For a majority of the populations, the International Waterbird Census provides an adequate framework to produce population size and trend estimates at regular and frequent intervals although inevitably with some time lag. However, improvements and enhanced coordination is needed for geese and sea ducks that cannot be well surveyed during core IWC counts because of their habitat use. In case of the core IWC counts, special attention is needed to strengthen the scheme not only in its core areas but also at the boundaries of each population’s geographic range, to account for potential range shifts related to climate change in the northern areas, in the Black Sea, Mediterranean (North East Adriatic, North Africa) and in the Sahelian zones. If possible, abundance monitoring should be complemented with demographic monitoring of age and sex ratios through direct observation and through wing samples, and estimation of survival rates through coordinated efforts across the EU on individual ringing and marking. It is also important to review delineations of flyway populations that represent reasonable units for management, in the light of improving knowledge of flyway connectivity patterns and with efforts to identify gaps and initiate research to fill those gaps. Improving flyway connectivity is not only important to improve harvest management, but also important for identification of key sites (and thus implementing the Birds Directive, AEWA, the Ramsar and Berne Conventions), but also in relation to help the management of diseases with high economic and conservation implications such as avian influenza.

The other core information need is harvest. With a few exceptions, the available data does not allow the European Commission, competent authorities of the Members States or other AEWA Contracting Parties to assess levels of harvest and their sustainability and, therefore, regulate hunting accordingly. In the absence of comprehensive information on total harvest across the flyway, Member States and Contracting Parties might permit levels of harvest that are collectively unsustainable or enforce unnecessary restrictions. Unsustainable harvests would contribute to the deterioration of the conservation status of the target population, while unnecessary regulations would undermine legitimate recreational interests and remove some incentives that could otherwise be harnessed to restore the population into favourable conservation status. For the sustainable management of migratory waterbird populations, all above-mentioned decision-making bodies should know the total harvest and understand its impact on the populations in the light of agreed conservation objectives. Unfortunately, the current system to collect harvest data is too fragmented and does not allow annual assessment of harvest levels that could be used to assess the sustainability of harvest and to take measures to flexibly adjust harvest levels to the required levels. It is important to emphasise that this is not only a problem for declining populations, but also for populations that grow rapidly and cause conflicts with human interests. A failure to understand that harvest levels are insufficient to adapt population to adequate levels can unnecessarily delay adjusting harvest levels or introducing additional control measures and can lead to higher costs of bringing back the population to the required level using resources that could be otherwise better used for addressing other conservation problems. **To overcome these limitations, we recommend that annual reporting on harvest levels of waterbird populations would be gradually introduced in the AEWA region. Ideally, reporting should be done annually and for all huntable species at national levels and collated internationally. If such general systems are not possible to implement in the short term, priorities should be given to declining populations of species listed on Annex II of the Birds Directive and on populations listed in Column A and B of**

AEWA because without adequate information on harvest at flyway level, it is not possible to fulfil the requirements of Art. 7.1 of the Birds Directive and of paragraph 2.1.1 and 2.1.2 of the AEWA Action Plan.

There are an increasing number of populations of species listed on Annex II of the Birds Directive that become subject of species action plans or management plans. Unfortunately, only a few of them (particularly the Svalbard pink-footed goose and the taiga bean goose plans) include an adaptive harvest management framework. Many others, including the EU management plans, do not even address harvest management or just simply introduce hunting ban without offering a clear perspective under what conditions that ban would be lifted. The Svalbard pink-footed goose International Species Management Plan clearly demonstrates the benefits of adaptive harvest management: it ensures a transparent process, adequate stakeholder involvement and dialogue, agreement on objectives and it leads to self-regulation amongst hunters. An added benefit of such an adaptive harvest management process is that, due to the built-in monitoring and adaptation processes, it can be implemented with rather limited information because the system allows learning and adjustments. **We propose that future AEWA and EU action plans and management plans for Annex II species should apply the principles of adaptive harvest management framework and make provisions for setting up adequate monitoring and information management systems and organisational structures to manage the decision-making process.** Beyond the legal ramifications of a failure to fulfil the requirements, the lack of harvest management practices that adequately account for scientific uncertainty and tolerance of risk may otherwise cause severe economic and cultural losses.

This report also shows that achieving sustainable management of waterbird populations has to be embedded in a coupled social-ecological system. We argue, therefore, that adaptive management of waterbirds at the flyway level needs to be built on the principles of both ecological and social theory to reflect stakeholder incentives and objectives. **We suggest that internationally coordinated management structures are established to facilitate dialogue, learning and communication between stakeholders with different objectives and cultural backgrounds.** These organizational improvements could incorporate and build upon existing international frameworks, e.g. AEWA Working Groups, or may necessitate new organizational structures. These may be employed to manage single species but it is envisaged that future organizational structures could concentrate on specific regional flyways, encompassing a range of species. Nevertheless, reframing international natural resource decision-making, taking into account social-ecological contexts, will facilitate the sustainable management of waterbirds, connecting and coordinating large-scale top-down (e.g. policy) and small-scale bottom-up (e.g. co-management of waterbirds) initiatives.

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