**Effects of stakeholder empowerment on crane population and agricultural production**

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# **Abstract**

Conflicts between opposing objectives of wildlife conservation and agriculture are increasing globally due to rising human food production and competition with wildlife over land use. Conservation conflicts are often complex and driven by variability and uncertainty in wildlife distribution and stakeholder wealth and power. To manage conflicts, empowering local stakeholders by decentralizing decisions and actions has been suggested to promote democratization and awareness of stakeholders. There is, however, a current gap in the understanding of how stakeholder empowerment (*e.g.,* farmers’ and managers’ practical, time or monetary resources) affects policy effectiveness. In this study, we apply an individual-based model of management strategy evaluation to simulate the conservation conflict surrounding protected and thriving common cranes (*Grus grus*) causing damage to agricultural production in Sweden and along the European flyways. We model the effect of farmer empowerment (*i.e.,* increasing budgets to effect populations and agricultural production) in four management scenarios, in which we manipulate the availability and cost of two actions farmers may take in response to crane presence on their land: non-lethal (scaring) or lethal (culling) control. We find that lower budgets lead to increases in population size, due to increased use of less costly scaring instead of shooting. Higher farmer budgets lead to increased population extinction risk. Intermediate budgets allow farmers to control the population size around the management target and limit impact on agricultural production to intermediate levels. Our study highlights that stakeholder empowerment and culling strategies based on the number of stakeholders, and particularly their power to implement effective actions, needs careful consideration and monitoring when setting management targets and strategies. Further, our results show that empowering individual farmers has the potential to contribute to conflict management and to balance agricultural with conservation objectives, but increased stakeholder involvement also requires careful planning and monitoring.

# **Introduction**

Conflicts between the objectives of wildlife conservation (*i.e.,* species protection, habitat restoration) and sustainable agriculture are increasing globally due to a rising human demand for food production and consequent competition with wildlife over land use (Henle et al., 2008; Redpath et al., 2013). Such conflicts have been identified as one of the major causes of failed management strategies and can thus result in negative impacts on conservation outcomes and stakeholders´ livelihoods and psychosocial wellbeing (Barua et al., 2013; Hodgson et al., 2019; Redpath et al., 2013). Due to their complex and dynamic nature, conflicts like these are often described as ‘wicked problems’, characterized and driven by the variation and uncertainty in, *e.g.,* wildlife distribution and inequity in stakeholder wealth and power (Bennett et al., 2017; Mason et al., 2018; Young et al., 2016). In the case of herbivore populations, conflicts often arise when wildlife forage on and cause damage to crops, and when management options to reduce this are limited due to protection of the species or social opposition towards mitigation interventions (Dickman, 2010). Particularly good examples of such ‘wicked’ conservation conflicts involve protected cranes (*Grus spp.*) and geese (*Branta, Anser spp*.), which currently have exponentially increasing populations causing negative impacts on agricultural land in Europe and North America, especially on land surrounding protected areas designated for those species and biodiversity in general (Cusack et al., 2018; Fox and Madsen, 2017; Nilsson et al., 2019). Negative impacts caused by wildlife are often managed by culling to reduce populations, under the assumption that agricultural loss will decrease accordingly (Montràs‐Janer et al., 2019). However, due to protection or practical limitations, non-lethal mitigation methods (*e.g.,* scaring, diversionary feeding, monetary compensation) are often required instead of culling (Cusack et al., 2018; EC, 2009; Owen, 1990). Yet, the management of many species, including cranes and geese remains problematic, in part due to time lags in centralized managers’ actions as a response to negative impact on agricultural production (Cusack et al., 2018). This may aggravate conflicts by weakening stakeholder trust and willingness to act in line with policy (Bennett et al., 2017; Cusack et al., 2019; Nilsson et al., 2018; Young et al., 2016). To manage conflicts like these, empowering local stakeholders and their expertise by decentralizing decisions and actions has been proposed to integrate complexity and democratization, into the management process (Mason et al., 2018; Raik et al., 2008; Redpath et al., 2017). However, there is a current gap in the understanding of how stakeholder empowerment (*i.e.,* practical, time or monetary resources) affects sustainability and extinction risk of the population as well as agricultural production. In the extreme, decentralizing decisions to farming stakeholders without providing budgets to enact actions may fuel frustration and conflict with policy makers and managers and lead to ineffective management; whereas unlimited farming stakeholder budgets may cause conflicts when conservation stakeholder objectives supersede management objectives (Mason et al., 2018; Redpath et al., 2013).

Decision making in natural resource management is traditionally assumed to be based on stakeholder objectives to maximize their returns in terms of livelihood given a limited set of actions, ability and budget to perform them (D. Hodgson et al., 2020; Milner-Gulland, 2011). For example, farmers aim to maximize their agricultural production given the land and labor they have, whereas managers might instead aim to keep wildlife populations within a range where long term viability is ensured given their ability to monitor the population and enforce their policies. In pursuing these aims, practical limitations (*e.g.,* available time and resources) will constrain the power of both farmers and managers to act, leading to trade-offs in decision-making. Game theory formally describes situations with such strategic actions, in which the consequences of an individual’s decisions are influenced by the decisions taken by others (Hauert et al., 2006). For example, the success of a manager’s decisions toward fulfilling conservation objectives through policy or economic incentives might be affected by the decisions of stakeholders to comply or not (Colyvan et al., 2011; Milner-Gulland, 2011; Redpath et al., 2018). Hence, while many existing models assume that policy will be enacted faithfully, this assumption might not always be realistic, such as under used culling quotas due to deficient number of hunters (Butterworth, 1999; Milne, 2018; Milner-Gulland, 2012).

Small perturbations in stakeholder behavior or policy in uncertain and dynamic systems might result in severe and unforeseen implications for the dynamics of the entire system, including both human behavior (*i.e.,* social tipping points) and increased extinction risk in the focal wildlife population (*i.e.,* ecological tipping points) (Heal and Kunreuther, 2012; Scheffer et al., 2012). Complexity and uncertainty make empirical evaluation of management outcomes (*i.e.,* agent responses) challenging. To account for uncertainty at multiple points in the management process, the management strategy evaluation (*i.e.,* MSE) framework has been developed (Smith, 1999). However, until now, MSE models have incorporated constant decision-making rules for single managers or farmers over time (Bunnefeld et al., 2013; Melbourne-Thomas et al., 2017; Milner-Gulland, 2011). A newly developed framework for generalized management strategy evaluation (*i.e.,* GMSE) includes scenarios that can include multiple independent stakeholders making individual decisions, as influenced by changes in resources, policy, and individual circumstance (Duthie et al., 2018).

In this study, we apply the management strategy evaluation framework using the GMSE R package to simulate the conservation conflict case of protected common cranes congregating in large numbers (*i.e.,* up to 26,000 ind.) at agricultural staging sites in Sweden, causing significant damage to agricultural production (inspected and compensated damage totals up to 200.000 Euros/year) (Montràs‐Janer et al., 2019; Nilsson et al., 2019). Cranes are protected in Annex I in the European Birds Directive and thus from culling to regulate the population. The directive states that the listed species’ survival and reproduction must be ensured in their distribution range, but allows for licensed lethal culling to mitigate negative impact on human livelihoods and when non-lethal damage preventive measures (*e.g.,* scaring, diversionary feeding) have been found unsuccessful (EC, 2009). Due to the protective legislation, no management targets are defined for the maximum populations size on either staging site or flyway level, but as these populations will likely continue to grow, so will the stakeholder demand for lethal or non-lethal crop damage preventive strategies and the severity of the conservation conflict (Fox and Madsen, 2017; Montràs‐Janer et al., 2019). By applying the MSE framework in this study, we aim to identify the effect of increasing stakeholder power (*i.e.,* decentralizing decisions to enact policy), on all aspects of the system, including the objectives to keep a viable wildlife population and sustainable agricultural production over time, and the implications for managing conflict. More specifically, we investigate how increasing the ability of individual stakeholders to enact decisions at the farm scale affects broader scale changes in expected crane population sizes and agricultural production in four possible management scenarios: *a.*) no management and no stakeholder power to affect cranes, *b*.) scaring and culling of cranes, with a management objective to allow the population to increase to an effectively high management target (*i.e.,* 100,000 ind.), *c*.) only culling allowed, but with an effectively high management target and *d*.) scaring and culling with a management objective to keep the population at a lower target (*i.e.,* 15,000 ind.) to lower the negative impact on agricultural production.

# **Model**

*2.1 The generalized management strategy evaluation (GMSE) framework*

GMSE simulates the management strategy evaluation process in a way that models goal-oriented behavior and spatial distribution of individual stakeholders, managers and wildlife using an individual-based (*i.e.,* agent-based) framework (Bunnefeld et al., 2011; Duthie et al., 2018). We used the package GMSE v0.6.0.4 in R to simulate the effect of increasing empowerment of stakeholders on the size of the crane population and agricultural production (Duthie et al., 2018). The model includes four sub-models operating in sequence over a single time step, in this study modelled as one year (see 2.1.3-2.1.6 & Fig. 1). For detailed information on definition of used parameters and modelling procedure, see Supplementary Material 1 & 2.

*2.1.1. Landscape properties*

Farmers, managers and cranes operate on an agricultural landscape (*L*), modelled as a torus of discrete cells, owned by individual farmers producing potentially variable yield of agricultural crop (*i.e.,* agricultural production). Each landscape cell thus has unique traits including farmer ownership, *x-y* location (*Lxy*), agricultural production, and density of cranes in each time step. Only one farmer can own a single *Lxy* cell, but any number of cranes can occupy a single cell.

In this study, *L* was constructed as a grid of 100 × 100 cells, representing a staging area of about 200 km2 (Nilsson et al., 2018), utilized by a total of 50 farmers earning the majority of their livelihood from agriculture (≥ 1 km2 per farmer; Holmer, 2016; The Swedish Board of Agriculture, 2017).

*2.1.1 Individuals*

GMSE models discrete individuals, which here include farmers, managers, and cranes. Each individual has potentially unique traits (*e.g.,* location, age and reproductive output for cranes, location of the farm and budget for farmers and managers), which potentially affect their behavior. Farmers own a fixed number of contiguous landscape cells (*L*, see above) on which they can perform one or more types of actions. Each farmer has a budget *Bf* for performing actions, which can be broadly interpreted to encompass one or more factors limiting the total number of actions possible for the farmer during a single time step such as time, money, or available equipment. Farmers attempt to maximize their total agricultural production *Yf* across all of the cells that they own. Managers do not own landscape cells, but instead set policy that affects the costs of performing actions for farmers (*Caction*). Managers themselves have a budget *Bm* for setting policy, which can be broadly interpreted to encompass the factors that limit the power of managers to make and enforce policy decisions. Managers either do not attempt to regulate crane population density, or attempt to maintain cranes at some target population density (*N†*) at every time step by setting policy (*i.e.,* costs for farmer actions).

To assess the reproductive output for cranes, we used empirical time series data (1989-2019) of maximum number of cranes staging at a typical autumn staging site (*i.e.,* Lake Hornborga) in Sweden (County Admistrative Board Västra Götaland, 2020; Nilsson, 2016) to replicate the exponential population growth and thus the reproductive output until year 2019 through simulations (*nrep* = 100). The starting population in these simulations was based on the empirical data of ca 3000 cranes from 1989 (County Admistrative Board Västra Götaland, 2020; Nilsson, 2016), and we used these data to find a birth rate that produced a reasonable approximation to the empirical curve (see Fig. S1 in Supplementary material 3). The terminal abundances from the 100 *nrep* were thereafter used as initial population sizes in simulations and expected offspring produced per adult individual (*Rb*) to simulate future population development from 2020 in the population sub model (see below). The expected number of offspring per individual based on the simulation output was *Rb* = 0.118. When used to parametrize the model, it was sampled from a Poisson distribution with a rate parameter equal to *Rb* for each crane of age of *Rar* > 3 (Nesbitt, 1992). Crane death occurred only as a consequence of culling or age *Rag* = 20 (unpubl. ringing data). Cranes are known to repeatedly return to and move within a smaller part of staging sites (Nilsson et al., 2018), and each crane is thus modelled to occupy a landscape cell within *Rm* = 4 cells in any direction of the cell last occupied in the previous time step *t* (*see 2.1.3 The population model*). Data to parameterize the individual effect of a crane on agricultural production are not known, but to model substantial but not complete loss of farmer’s agricultural production (Montràs‐Janer et al., 2019), we used 2 percent loss of remaining agricultural production *Yf* when an individual crane moves to and feeds in a cell *Lxy*. Individual cranes move to and feed on 10 cells of the landscape during one time step, hence potentially causing the maximum equivalent of a 20% loss of agricultural production across landscape cells. If multiple cranes arrive at the same cell in a time step (or a single crane arrives at the same cell multiple times), then we assume that (1-0.2)Nij of the agricultural production is consumed on the cell.

*2.1.3. The population model*

The first sub-model simulates the population dynamics of *Nt* cranes in time step *t*. Cranes arrive at a randomly selected cell within *Rm* cells of the one that they left in *t-1* with equal probability, then feed *Rfe* times. Between each feeding, cranes move to a cell within *Rm* cells in any direction from the currently occupied cell on the landscape randomly selected with equal probability. Cranes then give birth to young, then potentially die of old age (see *2.1.1. Landscape properties & 2.1.2 Individuals*).

Initial population size for each simulation was independently sampled from the distribution of simulated terminal abundances from the 100 *nrep* in years 1989-2019 (see 2.1., Fig. S1 in Supplementary material 3 and Supplementary material 1; min:16857, 1stQ: 17588, median: 18047, mean:18034, 3rdQ: 18484, max: 19680 ind.). From this initial population size, we simulated a further 30 years (2020-2049). Although it is inevitable that the European population of common cranes will eventually approach some carrying capacity *K*, there currently are no data to estimate *K* (Harris and Mirande, 2013); current exponential growth suggests that *K* might be quite high. We therefore allowed for unlimited exponential growth over the projected period. We model each time step *t* as a single year of migratory cranes arriving, using an agricultural landscape during staging, and departing at the end of the staging season (Nilsson et al., 2018).

*2.1.4 The observation model and how the manager observes*

The observation model uses a “virtual ecologist” approach (Zurell et al., 2010) to model manager observation of the crane population. We defined a monitoring method in which cranes are counted with complete accuracy on a subset of the landscape (*i.e*, 10 × 10 cells) and density is then extrapolated to estimate the crane population size assuming the same density over the entire landscape. Hence, estimated population size N̂will deviate from the true *N*.

*2.1.5 Manager decision making*

The manager sub model assesses N̂in relation to a management target, and sets policy by defining action costs (*Caction*; these may be conceptualized as, *e.g.,* time, practical or monetary costs for different actions farmers may take to affect agricultural production*,* including culling or scaring cranes). Managers use an evolutionary algorithm to set costs for each action; this algorithm models the heuristic process of the manager considering different potential policies and choosing one that will result in a crane population density nearer to the manager’s target (see Supplemental Material 1). The predicted effects of different farmer action costs are used in the fitness function of the evolutionary algorithm, and are calculated from the predicted effect that each action will have on crane abundance and the total number of actions predicted by the manager, as based on farmer actions in the previous time step (see Duthie et al., 2018 for details). Cost combinations that are predicted to result in crane population densities closer to the manager’s target have higher fitness and are therefore more likely to be chosen when the evolutionary algorithm has completed. The algorithm thereby heuristically chooses decisions that best reflect manager objectives by mimicking a process of evolution by natural selection (Hamblin, 2013, for detailed information about the GMSE R package see Duthie et al., 2018). Once the algorithm is completed, managers enact the cost of each available farmer action. The total budget for the manager was kept constant in the model (*i.e.,* *Bm* = 1000). See Supplemental Material 1 for details on parameterization of the evolutionary algorithm.

*2.1.6 Farmer decision making and actions*

In the stakeholder sub model, farmers implement actions with the objective to maximize their own agricultural production, constrained by the costs of individual actions as set by the manager’s policy and the farmers’ annual budgets (*Bf*). As with manager policy decisions, farmer decisions are chosen by running a single independent evolutionary algorithm for each farmer in each time step. Farmers recognize that the presence of cranes on a landscape cell has a negative effect on agricultural production, and will therefore use actions to try to effectively decrease the presence of cranes. Individual stakeholder decisions consequently affect cranes and agricultural production over multiple time steps. The possible actions that farmers can take to reduce impact on agricultural production include non-lethal scaring, one action of which causes one crane to randomly relocate to a new cell before damaging crops (note, this could potentially result in the crane resettling on another cell owned by the acting farmer), or culling, one action of which causes one crane to be completely removed from the landscape before damaging crops (see scenario *a-b*). All actions of all farmers are performed in a random order so that, for example, one farmer does not do all of their scaring or culling before another farmer and thereby cause differences in farmer’s agricultural production due to the order of farmer actions. Farmers can only take actions on land that they own.

*2.2 Management scenarios*

*2.2.1 Scenario A*

Scenario *a* is a null model for population size and mean agricultural production when no scaring or culling is conducted and the crane population can grow exponentially. Currently, there is a lack of management targets both nationally and for the whole population along the flyways, so this reflects the current lack of population regulation.

*2.2.2 Scenario B*

Scenario *b* models current management in Sweden, which allows for intensive scaring methods to reduce loss in agricultural production (specifically using scaring by the use of gas cannons, scare crows, flags). Very limited culling (*i.e.,* up to 15 individuals licensed to a minority of the farmers) is occasionally permitted by the managing authorities (EEA, n.d.). In our model, the strict licensing procedure for culling is represented by a high cost for culling for the farmer, unlike scaring, and was regulated in the simulations by setting *N†* = 100,000, *i.e.,* ensuring a condition where the manager would always set a high cost for culling. Non-lethal scaring may divert cranes from single cells at the landscape, but will not affect population size and a scenario only permitting scaring was therefore not considered.

*2.2.3 Scenario C*

Scenario *c* models a hypothetical scenario where although culling is expected to be costly (*N†* = 100,000, thus the manager is expected to keep costs of culling very high) there either is no alternative action possible, or alternative actions like scaring are prohibited or widely perceived to be ineffective and therefore never taken.

*2.2.4 Scenario D*

Scenario *d* models a situation where managers permit both scaring and culling, but *N†* = 15,000, meaning that managers attempt to keep the crane population at a size well below presumed carrying capacity. This allows for a more dynamic allocation of budget for both farmers and managers, which consequently mimics a trade-off between the objectives to sustain agricultural production while ensuring viability of the crane population.

*2.3 Performance metrics*

We simulated increasingly empowered farmers, running simulation for a range of available budgets *Bf* = {50, 100, …, 3950, 4000}, with 40 simulation replicates for each budget value across four management scenarios (*a-d*) over 30 years. We extracted selected performance metrics for each time step *t* (year 2020-2049); crane population size *N*, percentage of maximum agricultural production (*i.e.,* observed agricultural production (*Yf*)/maximum agricultural production) over the individual farmers’ cells (scenario *a-b)*, number of culled individuals per farmer (scenario *b-d*) and number of scaring actions performed per farmer (scenario *b & d*; Table 1). In dynamic socio-ecological systems like the one studied here, it may also be likely that empowerment of farmers (*i.e.,* budgets *Bf*) to perform actions varies among individuals at a given time step. To assess the effect of such potential variability in farmer’s budgets, we repeated simulations for each of the scenarios with budgets varying among individuals by *Bf* ± 50 in each time step.

# **Results**

We found that empowering farmers to enact actions affects management outcomes in terms of crane population size at staging sites and agricultural production at farm level (Fig. 2-4, & S2 in Supplementary material 3). Very low levels of farmer budget led to increases in the population because farmers chose scaring as a less budget demanding action, which helped individual farmers in the short terms by scaring birds off their land, but led to high population size across all farms in the landscape in the longer term (Fig 2-4c,d & S2 in Supplementary material 3). On the contrary, very high farmer budgets led to high extinction risk (up to 45%) of the population (Fig. 3b,d). Whereas simulations of intermediate budgets allowed farmers to control the population size around the management target and kept the impact on agricultural production to intermediary levels. (Fig.2-4d)

*3.1 Scenario A*

S*cenario a* illustrates a long-term and exponential increase of the crane population from year 2020 (mean *N* = 19123) to 2049 (mean *N* =109298; Fig.2a). Changing farmer budget did not affect expected population size or agricultural production in at *t* + 5 (year 2024) (Fig. 3a & 4a) due to lack of management and inability to scare or cull (*i.e.,* scaring and/or culling and targeted population (*N†>N*)).

*3.2 Scenario B*

Results from *scenario b* demonstrate a long-term and exponential population increase over years in the same way as in scenario a (year 2021 mean *N*: 19018 ind. and for 2050 mean *N*: 102275; Fig. 2a, b). As the management target was always higher than the realized population (*i.e.,* *N†* > *N* at year 2024), the manager set the costs for culling high relative to scaring (mean *Cculling* = 7.1 × *Cscaring*) to sustain the population, which incentivized the farmers to allocate their budget to scaring instead of culling in year 2024 (Fig. S3 in Supplementary material 3). Nevertheless, occasional and limited culling (mean culled cranes per farmer: 0.004) occurred as farmer budget increased. Consequently, the population size in year 2024 decreased slightly relative to scenario a (mean *N* from 22691 to 18201 ind; Fig. 3b.) and mean agricultural production increased from 64.9% to 71.0% of total expected production (Fig.4b), within the range of increasing farmer budget (*Bf*: 50-4000).

*3.3 Scenario C*

In *scenario c,* the population increased exponentially over years, given intermediate farmer budgets (*Bf*:1000, for 2021 mean *N*= 18931, and for 2050 mean *N*= 67496, Fig.2c.). However, the population growth was slower compared to scenario *a* and *b* (Fig. 2a, b, c). Further, for a given time step (*t* > 5) increasing farmer budgets caused the population size in 2024 to decline (e.g. mean *N* = 23392 for *Bf*: 50 and *N*= 13120 for *Bf*: 4000; Fig. 3c) as a consequence of increased number of culled cranes per farmer (*i.e*., 0-36 cranes for *Bf*: 50-4000; Fig. S4c in Supplementary Material 3). As a result of population limiting effects, the agricultural production increased from 64.7 % to 78.4 % along the same range of farmers budgets (Fig. 4c). Since culling is the only available but yet costly action, stakeholders will have no alternative to culling when budget allows. This causes stakeholders to take action simultaneously when budgets exceed the cost of culling (*Bf* > *Caction*; Fig. S4c in Supplementary material 3). However, when adding variability to the farmer budget, the small threshold effects in populations size and agricultural production smooths out, as the farmer budgets exceeds their costs and incentivize actions at varying times (Fig. S5 in Supplementary material 3).

*3.4 Scenario D*

In *scenario d*, the population approached the management target of 15,000 cranes with increasing farmer budgets and aligned with the targeted population size for a limited intermediate budget range of (*Bf*: 550-850). However, the population declined to below the targeted population size when farmer budgets further increased (*Bf* > 850) as result of stakeholder power overriding manager ability to set costs to minimize culling (Fig. 2d & 4d). The lowered manager target, compared to *scenario a-c*, caused the manager to dynamically adjust the relative costs of scaring and culling depending on the size of the current population relative to the target. When current population size was greater than the defined target, the managers incentivized culling by lowering the culling costs relative to scaring, whereas the opposite occurred if the population is lower than the defined target. This is also illustrated by a continuously increasing number of culled cranes per farmer (*i.e.,* up to 35 cranes per farmer; Fig. S4d in Supplementary material 3) until the population aligns with the management target (*Bf*: 550-850; Fig. 2d), and occasionally with even greater number of culled cranes per farmer (maximum 57 ind; Fig. S4d in Supplementary material 3) as farmer budget outstripped the power of managers to set culling cost. Mean agricultural production varies between 73.9-75.9% when the population aligned with the targeted population size, but increased further up to 95.3% as the population declines along the range of further increasing farmer budgets (*Bf*: 900-4000) and below the targeted population size with an extinction risk of 45 % (Fig. 3d).

# ***Discussion***

*4.1 Stakeholder empowerment in multi-objective management*

In order to manage conservation conflicts, collaboration between stakeholders and managing authorities to decide on trade-offs between multiple objectives (*e.g.,* sustaining crane population and agricultural production) and providing local stakeholders with power to enact management, have been addressed as critical, and potentially more important than merely reducing negative impact from wildlife (Mason et al., 2018; Redpath et al., 2017, 2013). However, predicting the effects of providing stakeholders with power on the likelihood of pursuing management objectives (e.g., high extinction risk or the crane population being far from the management target and low agricultural production) have generally been over-looked when predicting management outcomes and risk of conflict from models (but see Bunnefeld et al., 2017; Milner-Gulland, 2011). Our findings demonstrate that empowering a large number of stakeholders comes with the challenge of finding a delicate balance of the degree of empowerment. Very high farmer stakeholder power means the wildlife population is at an increased risk of extinction (e.g., 45% year 2024 in scenario *d*) when lethal control is an option whereas very low stakeholder power means high wildlife populations and extensive negative impact on agricultural production. Our study shows that knowledge and management of stakeholder power and actions is needed to manage a sensitive trade-off between wildlife conservation and agriculture. Furthermore, our study shows that by modelling the interaction between two groups of stakeholders (manager and farmers) with potentially opposing views, we contribute to our understanding of the complexity as well as challenges and risks of stakeholder empowerment and decentralization of decision making.

As the protected crane population along the European flyways likely will continue to increase, so will the negative impact on agricultural production, and trade-offs in management objectives between agricultural production and a sustainable crane population, affecting associated conservation conflict. A multiple-objective management will require interventions to regulate the population size with sustainable culling strategies, careful monitoring and modelling efforts to predict the likely impacts of alternative strategies (Cusack et al., 2018; Johnson et al., 2014). Our modelling demonstrates that a large community of farmers managing the crane population in a coordinated fashion may effectively regulate the population, and thus maintain (or even increase) agricultural production (scenario c-d). The specifics of the quota system set by the manager will have an effect on population size. For example, culling quotas of a limited number of cranes to all individual farmers (as opposed to, e.g., higher quotas to a very small number of individuals, as currently is the case) may be an approach to increase equitable distribution of individual power (Redpath et al., 2017). However, our simulations also highlight that the total number of licenses to cull cranes needs to be carefully considered to maintain population sustainability (Fig. 2c,d & S4 in Supplementary material 3). This is true especially if no other management actions (e.g. scaring, diversionary fields) are possible, as even small increases of a few cranes to cull per stakeholder may cause the population growth rate to decline (scenario c & d). Today’s license system in Sweden is based on very limited quotas from the county administrative boards (up to 10 cranes) to a minority of the farmers (EEA, n.d.), which based on our results likely have insignificant effects on either population size or agricultural production. Yet, our findings show that the extent to which farmers´ impact on the crane population and agricultural production can be significant is dependent on the power given to the farmers. This could be an effective way to perform management in line with policy and to decrease practical or monetary costs for farmers. It could for example include compensation schemes for farmers to pay for labor connected to culling or scaring actions, provision of scaring devices, decoys or hides for culling, and coordination and help by employed personnel to scare and cull cranes (Hake et al., 2010; Nilsson et al., 2018). Our study exemplifies that if licensed culling would be permitted (*i.e.,* relocated from annex I to annex II in the EU Birds directive; EC, 2009), extensive stakeholder effort would be needed to reduce the population to a lower management target (e.g., >25 culled individuals per farmer after 5 years in our model). Accordingly, our findings have implications for the evidence-informed flyway management plans implemented for several thriving goose species under the United Nations African Eurasian Waterbird Agreement (AEWA) (Madsen et al., 2017; Stroud et al., 2017). These management plans consist of agreed population targets, to be delivered via adaptive management, population monitoring, population estimation and country-specific culling quotas (Madsen et al., 2017). Our study highlights that stakeholder empowerment and culling strategies based on the number of stakeholders, and particularly their power to implement effective actions, needs careful consideration and monitoring when setting population targets and decentralized policy (Baynham-Herd et al., 2018; Mason et al., 2017; James H Williams and Madsen, 2013).

*4.2 The Management Strategy Evaluation framework to model socio-ecological systems*

Our findings demonstrate how individual-based modelling and the management strategy evaluation framework can be used to investigate the effects of manager’s and stakeholders’ interactive decisions in management of natural resources and ‘wicked’ conservation conflicts in complex and uncertain systems, such as for a protected and increasing crane population causing negative impact on agricultural production (Bunnefeld et al., 2011; Duthie et al., 2018; Smith, 1999). Until now, MSE models have used constant decision-making rules for single stakeholders over time (Bunnefeld et al., 2013; Melbourne-Thomas et al., 2017; Milner-Gulland, 2011), whereas we used GMSE to simulate scenarios including a large number of individual stakeholders taking decisions independently, as influenced by changes in the wildlife population, agricultural production and policy set by the manager. GMSE is freely available as a package in the R environment and all model code is accessible, further allowing the development of increased use of the model and the empowerment of all stakeholders through investment in communication and co-development of the model (Duthie et al., 2018).

Socio-ecological and individual-based models like these simplify empirical systems to investigate key concepts and clarify theory, and inevitable brings complex trade-offs between the extent of realism and integration of knowledge (Schlüter et al., 2019). Our model necessarily makes some restrictive assumptions about crane ecology and management due to limited access to empirical data for model parameterization. For example, we do not have data on individual cranes´ impact on agricultural production, or on the exact economic consequences for farmers, which likely is influenced by e.g. market prices, weather, and timing of harvest (Montràs‐Janer et al., 2019; Nilsson et al., 2016). Further, socio-ecological and individual-based models are based on the assumption that human decision-making is bounded rationale (Schlüter et al., 2019). In our model, farmers and managers are assumed to take decisions relating to a single objective (*i.e.,* farmers to maximize production, managers to minimize the difference between current and targeted crane population size). Instead, in real-world situations, individual managers and farmers with diverse norms are likely to adaptively make trade-offs between multiple objectives over time and in relation to decisions taken by other stakeholders (Schill et al., 2019). In the context of cranes and farming, an example could be farmers potentially tolerating a certain number of cranes and thus negative impact on their agricultural production to sustain biodiversity, given certain management conditions. The mechanisms for such stakeholder multi-objective trade-offs are not very well understood to date, and require further empirical and theoretical study (Bunnefeld et al., 2017; Schill et al., 2019)

*4.3 Conclusions*

A large number of stakeholders, e.g., farmers and managers, naturally causes complications to optimal trade-offs in decision making in natural resource management, and small changes in stakeholder behavior or policy are found to potentially have large scale implications for the dynamics of entire socio-ecological systems (Heal and Kunreuther, 2012; Scheffer et al., 2012). To advance previous modelling of the MSE framework, we used the GMSE model to simulate and predict the long-term effects of empowering multiple farmers to dynamically respond to policy, the crane population and impact on agricultural production under uncertainty. Our findings demonstrate that empowering a large number of farming stakeholders to pursue their management objectives comes with the challenge of finding a sensitive balance of the degree of empowerment. Not providing stakeholders with budgets to enact actions may lead to frustration, ineffective management and risk of conflict; whereas unlimited farming stakeholder budgets may cause conflicts when conservation stakeholder objectives overrides management objectives (Mason et al., 2018; Redpath et al., 2013). This suggests that collaboration between managing authorities and stakeholders may be critical to agree on and act in line with multi-objective management targets. Carefully considered levels of stakeholder empowerment may so enhance democratization and trust between parties with the aim to manage ‘wicked’ conservation conflicts (Mason et al., 2018; Redpath et al., 2017; James H. Williams and Madsen, 2013).

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**Figures and tables**

Table 1. The management scenarios simulated over 40 time steps in the GMSE v0.6.0.4 R package (Duthie et al., 2018). The management target is set by the manager and is given in number of cranes on the simulated landscape (*i.e.,* staging site).

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Scenario** | **Target** | **Culling** | **Scaring** | **Addressed management** |
| A | NA | No | No | No management w. objective to sustain an increasing crane pop |
| B | NA | Yes | Yes | Scaring, culling, w. objective to sustain an increasing crane pop |
| C | NA | Yes | No | Culling, w. objective to sustain an increasing high crane pop |
| D | 15,000 | Yes | Yes | Scaring, w. objective to sustain crane pop and agri.prod |

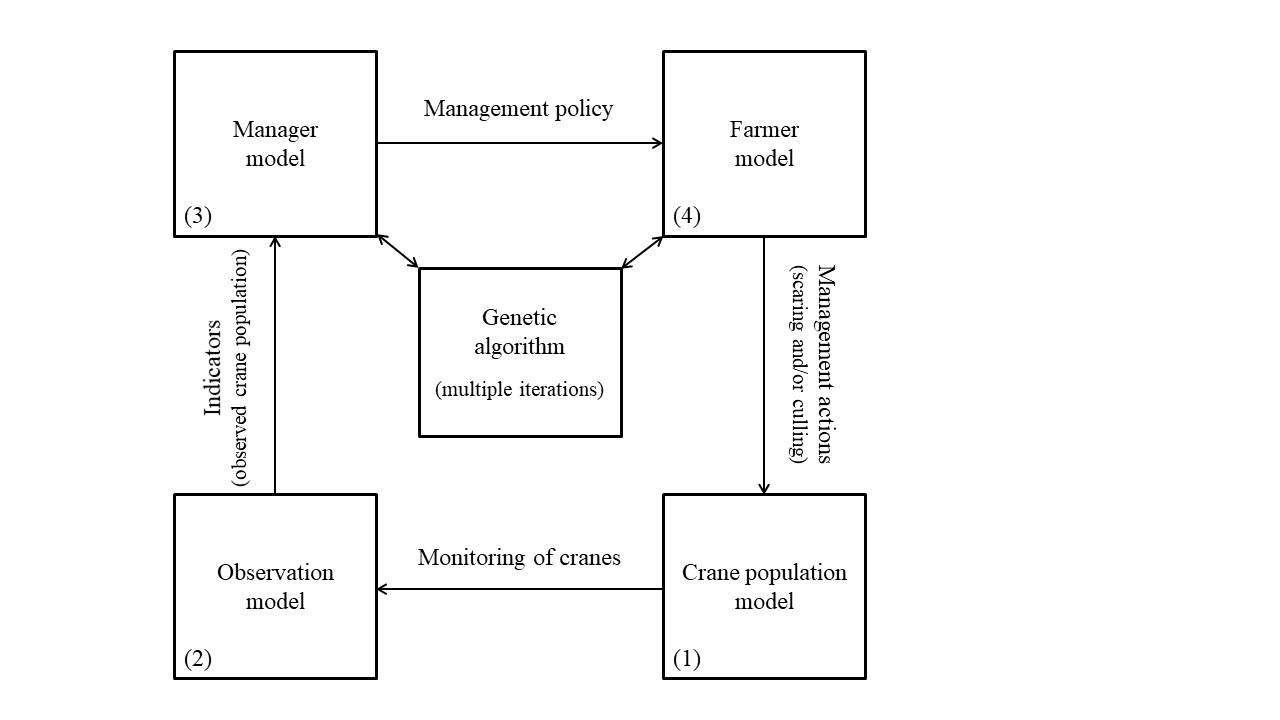


Figure 1. The concept of the generalized management strategy evaluation modelling (GMSE) framework. The model consists of four sub model (see number 1-4) operating in sequence for each time step.

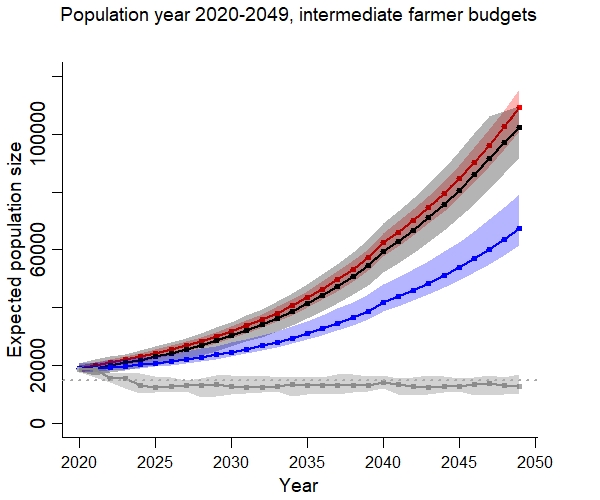


Figure 2. Population size of common cranes over 30 years, given four different management scenarios and given intermediate farmer budgets (*B*f: 1000) for 50 stakeholders. The management scenarios are: a.) no management (red lines), b.) scaring and culling, without a realized management target (black lines), c.) only culling, without a realized management target (blue lines), d.)  scaring and culling (grey lines), with a management target of 15,000 cranes (dotted grey line). The mean (points joined by lines), minimum and maximum (shaded areas) expected population sizes are based on data from 40 model simulations.

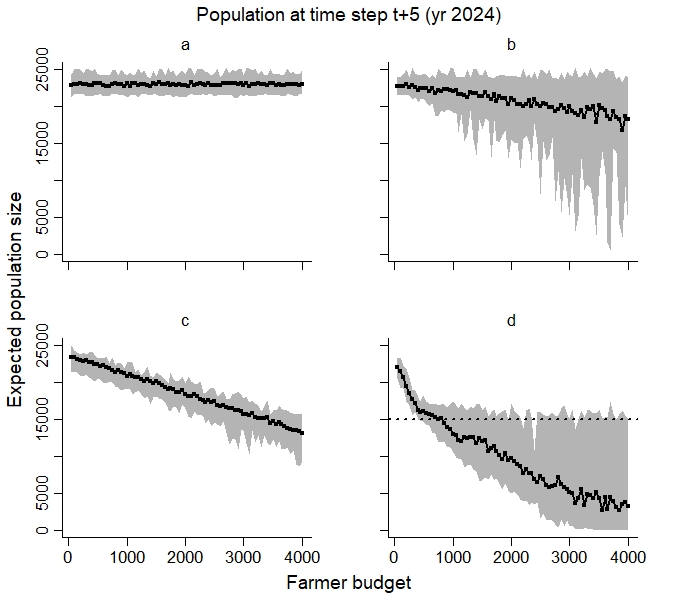


Figure 3. Effect of farmer budgets on expected population size of common cranes at year 2024 year in four different management scenarios; a.) no management, b.) scaring and culling, without a realized management target, c.) only culling, without a realized management target, d.)  scaring and culling, with a management target of 15,000 cranes (dotted black line). The mean (black line), minimum and maximum (grey shaded areas) expected population sizes are based on the simulation output data at year 2024, given 50 stakeholders and are produced from 40 model simulations.

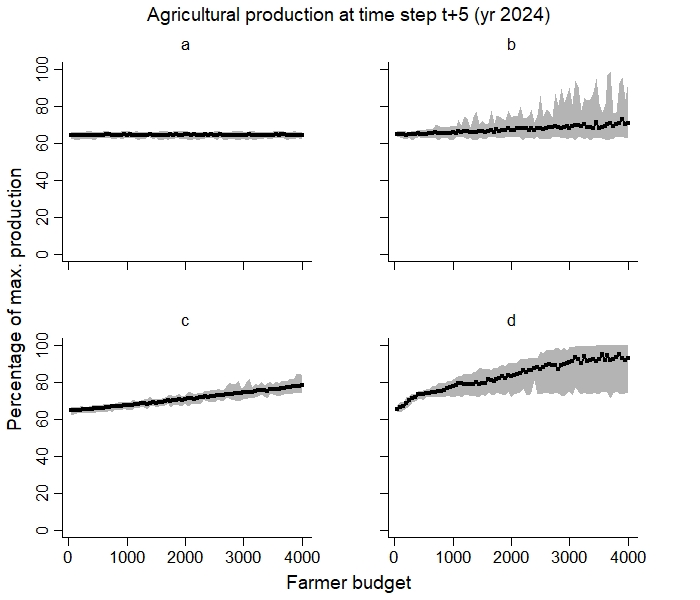


Figure 4. Effect of farmer budget on mean agricultural production per farmers’ land at staging sites of common cranes in four different management scenarios at generation 30, 1.) no management, 2.) scaring and culling, without a realized management target, 3.) only culling, without a realized management target 4.)  scaring and culling, with a management target of 15,000 cranes. The mean (black line), minimum and maximum (grey shaded areas) mean agricultural production per farmers’ land are based on the simulation output data at year 2024, given 50 stakeholders and produced from 40 model simulations.