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Developing spatio-temporal models for landscape-scale pesticide ERA

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1. Dansk Sammenfatning

I denne rapport undersøger vi, hvordan risikovurdering af pesticider kan foretages på landskabsniveau for hele bestande af forskellige organismer ved hjælp af simuleringsteknikker. Risikovurderingsprocessen er kendt som *Environmental Risk Assessment* (*ERA*) og denne forkortelse vil blive brugt i det følgende.

I de eksisterende regelsæt (EU guidance documentation) er der ikke specifikke krav om ERA på bestands- eller landskabsniveau, men ifølge den gældende lovgivning (*EC regulation No 1107/2009 (21 October 2009))* fremgår det, at det overordnede mål er, at arterne er beskyttede på bestandsniveau og at økosystemydelserne ikke forringes. Derfor arbejder den europæiske Food Safety Authority (EFSA) på en strategi for 2020, der vil komme til at indeholde en anbefaling om at ERA foretages på landskabsniveau.

Vi giver eksempler på sådanne ERA på landskabsniveau for fugle, pattedyr og terrestriske invertebrater ved brug af eksisterende modeller for hare (*Lepus Europeaus*), sanglærke (*Alauda arvensis*) og en løbebille (*Bembidion lampros*). Herudover har vi i dette projekt udviklet en ny model for stor vandsalamander (*Triturus cristatus*). Padder er endnu ikke omfattet af krav til ERA, da det antages, at de dækkes af kravene til fugle og pattedyr. Denne beslutning er dog under overvejelse i EFSA og derfor er Stor vandsalamander en oplagt kandidat for fremtidige ERA på landskabsniveau.

Vi baserer risikovurderingerne på simulerede ændringer i bestandene for de undersøgte modelorganismer. Vi ser dels på den måde dyrene fordeler sig i landskabet (dispersion) og tætheden af individer, hvor de findes. Disse to mål kombineres i et såkaldt *AOR-index*, der er et todimensionelt indeks over de relative ændringer i henholdsvis fordeling og tæthed i forhold til et baseline scenarie. Herudover bruges den relative bestands-vækstrate (*'relative population growth rate'* eller *pgr*) som et mål for en given bestands mulighed for at overleve på langt sigt.

Alle scenarier blev kørt i ALMaSS, der er et agent-baseret simuleringssystem og som kan simulere effekten af pesticider i forskellige landskabs-scenarier. Da dette projekt omfattede fire modelorganismer, mange scenarier og replikater, og da hver enkelt model krævede megen computerkraft, blev modellerne kørt på et stort Linux-cluster på Centre for Scientific Computing ved Aarhus Universitet (http://www.cscaa.dk)

Vandsalamander modellen er nyudviklet i forbindelse med dette projekt og den beskrives derfor mere detaljeret. I denne model simuleres æg, larver, juvenile, samt voksne hanner og hunner som selvstændige individer. Modellen blev udviklet med et såkaldt mønster-orienteret tilgang og blev efterfølgende testet for sensitivitet og statistisk usikkerhed. Den færdige model var i stand til at simulere de store udsving i bestandsstørrelse og tæthed, der er beskrevet i litteraturen og viste bl.a., at salamanderens terrestriske stadier er særligt følsomme for dødelighed forårsaget af vejrliget. Vi vurderer, at modellen er tilstrækkelig præcis til at kunne anvendes i forbindelse med alle standardscenarier, men diskuterer de begrænsninger, der er fundet, og hvordan den kan videreudvikles.

I dette projekt brugte vi en række standardscenarier, der blev anvendt for alle fire arter og med de samme analyser. Scenarierne blev udviklet således, at de var sammenlignelige mht. effekt. Det betyder, at doseringerne er specifikke og tager højde for hver enkelt arts følsomhed. Effekten på de forskellige organismer blev derefter fortolket individuelt. Udover standardscenarierne konstruerede vi også en række artsspecifikke scenarier for at kaste lys på særlige problemstillinger ved anvendelsen af denne type ERA.

Standardscenarierne blev brugt til at demonstrere effekten af landskabsstruktur, arealanvendelsen og toksiciteten af de anvendte pesticider. De artsspecifikke scenarier blev brugt med forskellige formål for øje. For haren undersøgte vi, hvilken effekt landskabsstruktur og landbrugspraksis har på ERA resultaterne. Løbebillen blev brugt som et eksempel på en art med flere livsstadier og her blev de specifikke scenarier brugt til at vurdere konsekvenserne af at modellere disse stadier individuelt eller kombineret i én og samme simulering. Sanglærken blev brugt til

at teste, hvorvidt modellen kan simulere de konkrete biologiske effekter af, at et pesticid giver øget risiko for at lærken lægger æg med tyndere skaller. Vandsalamander-modellen blev brugt til at demonstrere, hvordan laboratorietest af toksicitet kan anvendes til at forudsige de afledte effekter af pesticidanvendelsen på bestandsniveau. Denne model blev også brugt til at evaluere, hvad det betyder for simuleringen at introducere tilfældig variation i vandhullernes kvalitet.
Simuleringerne viste, at alle arter påvirkes af landskabsstrukturen, men at effekten varierer mellem arterne. Alle arterne responderede som forventet på omfanget af pesticidanvendelsen og toksiciteten af stofferne. For sanglærken var omfanget af det totale areal påvirket af pesticidanvendelsen alene i stand til at forudsige en bestandseffekt. For de øvrige arter kunne areal og frekvens af pesticidanvendelsen og et index for behandlingerne bruges prediktivt. Vi fandt dog enkelte landskaber, hvor det ikke var muligt at påvise en sammenhæng mellem pesticidanvendelsen og bestandseffekter på hare og sanglærke. For hare, sanglærke og løbebillen viste modellen en risiko for at et fald i bestandsstørrelse over en længere tidshorisont.

De artsspecifikke scenarier viste, at resultaterne af en ERA for hare påvirkes af landbrugspraksis og dyrkningsmønster og landskabsstruktur (i forholdet 2:1). For sanglærken viste resultaterne, at den præcise timing af pesticidanvendelsen har stor betydning i de scenarier, hvor fuglene lægger æg med tyndere skaller. Her kunne modellen ydermere vise, hvordan særlige biologiske træk ved sanglærken i mindre grad kan kompensere for disse effekter. Løbebille-modellen viste, at en ERA giver et mere konservativt skøn, hvis livsstadierne modelleres individuelt end hvis de kombineres i en enkelt simulering. Endeligt viste vandsalamander-modellen, at det at indføre en stokastisk variabel for levesteds-kvalitet ikke ændrede resultaterne af vurderingen signifikant. Denne model viste også, at der er en risiko for at overse langtidseffekter af pesticidpåvirkning, hvis der kun overvåges en kort årrække.

Resultaterne fra dette projekt bekræfter, at landskabets struktur og dynamik har en meget stor indflydelse på en ERA og at arternes individuelle og unikke biologi og adfærd medfører reproducérbare, men usammenlignelige vurderinger, der er resultatet af individernes samspil med og i det konkrete scenarie. Det betyder, at den i ERA almindeligt anvendte '*worst case*', er svær at identificere generelt, men må udvikles for enhver unik kombination af art, landskab og landbrugspraksis.

For at gøre det muligt at bruge ERA på landskabs-niveau, vil det derfor være nødvendigt at udvikle en række artsmodeller, der er testede og generelt accepterede. Det er ligeledes nødvendigt at udvikle en række standardlandskaber, der repræsenterer variationen af landskabsstruktur, arealanvendelse og alle potentielle former for god landbrugsmæssig praksis. Dette vil kræve samarbejde og udvikling af fælles standarder for model-interface og scenarieudvikling, men vil gøre det muligt at foretage præcise ERA, der lever op til intentionerne i lovgivningen.

2. Summary

This project investigates various aspects of landscape-scale population-level environmental risk assessment (ERA) for the purposes of regulatory risk assessment for pesticides at EU and member state level.

Under current EU guidance documentation there is no provision for landscape-scale and population-level ERA. However, the current legislation (EC regulation No 1107/2009 (21 October 2009)) requires protection at the population level, as well as for ecosystem services, and current European Food Safety Authority (EFSA) and EFSA strategy for 2020 firmly introduce the concept of landscape-level risk assessment. Therefore, this type of ERA will be a reality at some point in the near future.

We provide examples of landscape-scale population-level ERA for birds and mammals, and terrestrial invertebrates using existing models for the European Brown Hare (*Lepus europeaus*), the Skylark (*Alauda arvensis*), and a carabid beetle (*Bembidion lampros*). We also develop a model for Great Crested Newt (*Triturus cristatus*). Amphibians are not currently addressed in regulatory ERA but are assumed to be covered by bird and mammal risk assessment. This situation is now under review by EFSA, therefore the newt is a likely candidate for landscape-level risk assessment in the future.

Endpoints considered for the risk assessment were changes in overall population size, changes in landscape occupancy (dispersion) and density of animals where present. These latter two were combined in the AOR-Index. A measure of relative population growth rate (relative pgr) was also used to assess the potential for long-term population decline.
All simulations were run using the models from ALMaSS, an agent-based simulation system designed for assessing comparative landscape scenarios. Simulations were run using a powerful Linux cluster due to the very large number of scenarios and replicates, and the very long run times of individual runs.

The newt model was developed as an agent-based model within ALMaSS. Eggs, larvae, juveniles, males and females were simulated as individuals. The model was developed using a pattern-oriented modelling approach and a subsequent sensitivity and uncertainty analysis was carried out. The final model was particularly sensitive to weather induced mortality of terrestrial stages. It was able to replicate the large between-year fluctuations in population density as well as survivorship and density of newts as described in literature. The model performed acceptably and was used for the standard and species-specific scenarios. However, limitations and recommendations for future developed were identified.

The approach taken was to create a set of standard scenarios that were applied to all species, together with a standard set of analyses. The scenarios were developed in such a way that they were directly comparable between species in terms of the application rate and toxicity that triggered an effect per individual if exposed. The effect triggered was then interpreted differently for each species (e.g. mortality for beetles, reproductive effects for skylarks). In addition to the standard scenarios a set of specially constructed species-specific scenarios were developed to highlight a particular aspect or aspects of using this type of ERA.

Standard scenarios evaluated the effect of landscape structure, scale of use and toxicity. Species-specific scenarios for the hare were used to evaluate the effect of landscape structure and the farming it contains on the results of the ERA. The beetle species-specific scenarios investigated the consequence of addressing risks to different life-stages separately or together in the

same simulation. The skylark was used as an example of evaluating the biological relevance of a laboratory test, using an eggshell thinning scenario, and the effect of timing of application. The newt was used to demonstrate how laboratory toxicity tests can be translated to population-level impacts when setting specific protection goals. It was also used to evaluate the impact of introducing stochasticity of pond quality into the scenario to account for the fact that variability in pond quality was not mechanistically modelled.

All species were affected by landscape, but the effect varied between species, and there was little in the way of correlated changes in changes in the ERA because of landscape between species. Generally, all species did respond as expected in terms of increasing scale of use and toxicity. Scale of use was a reasonable predictor of effect in skylarks, but scale of use including a treatment index was a better predictor of effect in the other three species. In some landscapes scale of use and treatment index did not correlated well with effect for hares and skylarks. All species except the newt showed the potential for long-term (over decades) decline in population size as a result of pesticide application.

Species-specific scenario results showed that for hares the variation in ERA as a result of landscape context was primarily caused by farming in a ratio approximately 2:1 with landscape structure. Timing of application was critical in skylark eggshell thinning scenarios, and ecological details were shown to be important in the species ability to compensate for pesticide induced reproductive losses. In the beetle, separate assessment of effects on larvae and adults was more conservative than combining these in a single assessment. The newt scenarios indicated that stochastic pond quality did not significantly change effects over a slightly more conservative assumption of global optimal pond quality. They also showed that there was potential for small long-term population effects to go unnoticed in short-term studies.

The main result of the simulations confirms that landscape context has a very large influence on the results of an ERA. In addition, the results were not transferable between species, and species ecology interacted with landscape context to create reproducible but unpredictable variation in ERA under different conditions. As a consequence, the concept typically used in ERA of realistic worst-case scenario cannot be generally applied, but must be created for each specific set of contexts.

To develop landscape-level risk assessment as a reliable tool in regulatory risk assessment it will therefore be necessary to create a set of species models that are agreed and tested. These models should have standard interfaces to standard models. However, it is important that the regulatory scenarios include a wide range of landscapes and all potential GAPs simulated. Standard landscapes should therefore be developed, which will be facilitated by the methods developed under this project.

We recommend that the regulatory scenario used for landscape-scale population-level ERA will need to include a specific consideration of the landscape structures, farming, and weather and a justification for the form of inclusion of individual toxicity.

3. Introduction

There is a clear need to evaluate the impact of pesticides on non-target organisms, not least due to the legal requirements for regulating plant protection products and evaluating the risks they entail to non-target organisms which are set down in the EC regulation No 1107/2009 (21 October 2009). There is, however, a discrepancy between the methods used to assess risk as defined by the EU directives, and our ecological knowledge-based expectations of the impacts of pesticides. This discrepancy is based on the fact that the administrative system requires a clear, objective and simple approach to pesticide regulation, resulting in current requirements to assess a single product in isolation (i.e. no multiple stressors), largely ignoring issues such as scale of use, year-on-year application, spatial-dynamic, and population processes. We know that these factors both singly and interacting can cause significant changes to the estimate of impact when using pesticides (Dalkvist, Topping et al. 2009, Topping and Lagisz 2012, Forbes and Calow 2013).

One way to improve the realism and therefore accuracy of the risk assessment is by using more detailed simulation modelling (Forbes and Calow 2012, Topping, Kjaer et al. 2014). However, application of simulation models for pesticide risk assessment is in its infancy and to date only two terrestrial higher tier regulatory risk assessments for pesticides for single substances have been carried out at landscape scales (both using the ALMaSS vole model). Thus, the full benefit of these modelling systems has not been utilized. Modern agent-based models (ABMs) are characterised by their facility to integrate many factors varying in time and space and thus allow us to consider highly realistic model systems capable of integrating timing and use of pesticide with ecology and behaviour of the animals. These systems also include local interactions and feedback loops thus generating dynamics mechanistically which have previously only been simulated as fixed parameters (e.g. as incidence functions in metapopulation models).

3.1 Reading guide for the report

Because there is currently little work done on landscape-scale population ERA there is no particular area of focus or set of important questions to answer. Rather we are at a stage of demonstration of potential and generation of experience and understanding of the uses, constraints and potentials of this approach. As such, this project sought to demonstrate a range of issues and factors to consider when developing these approaches. This was achieved through use of examples arising from the application of landscape-scale population ERA to a set of standardised scenarios for terrestrial non-target organisms.

This study utilizes three existing models as well as develops one new model that can be used for risk assessment. To help in following the report it is useful to be aware of the overall structure. Here the idea is that each species model is used in a set of standard scenarios that can be directly compared. The standard scenarios were designed to highlight effects of changing toxicity, scale of use and landscape structure. Then for each species we have created new scenarios that highlight a specific factor to be considered in landscape-scale population-level environmental risk assessment for pesticides.
An overview of the methods is presented in the main text but many details of the methods are separately described in Appendices. The reason for this is that otherwise the report would be extremely methods heavy and most details are not necessary to follow the scenarios.

The results are presented per species, including the species-specific scenarios, there then follows a combined analysis of effects across species.

3.2 Current terrestrial Environmental Risk Assessment (ERA) for pesticides

The current ERA is a tier-based assessment with in the EC regulation No 1107/2009 (21 October 2009) as the legal basis. However, there is no single ERA, rather a set of ERAs are carried out covering different groups of organisms. To enable national authorities and the European Food Safety Authority (EFSA) to carry out these ERAs there is a number of guidance documents that have been adopted by the EU Commission and form the basis for evaluating whether ERAs submitted by industry in support of registration of pesticides have been carried out correctly. These guidance documents are under a programme of cyclical updating carried out by EFSA in conjunction with the EU Commission. The typical cycle is that EFSA receives a mandate from the Commission to evaluate the need to update specific guidance, then proceeds to create a scientific opinion addressing the ERA needs. This takes approximately 2 years and then if accepted by the Commission there will be a delay (may be another 2 years or more) before a mandate is received to update the guidance. As such, there is considerable delay between updates for different groups and the future ERA in terms of the scientific opinions is signposted long before new guidance comes into force. Currently, this delay is even longer since the trigger for the mandate to produce guidance documentation is the setting of the specific protection goals by the Commission. This has been delayed; therefore, no new guidance is currently under development.

As a result most current guidance was actually developed under EU Directive 91/414/EEC where the protection goal was effectively specified at the individual level, and terms like population, landscape, and multiple-stressors were not explicitly considered, rather effects related to these factors were assumed to be ‘covered’ by the application of assessment factors. The new regulation from 2009 changes the protection goal to be the population level for most groups, and recent EFSA scientific opinions (e.g. EFSA Panel on Plant Protection Products and their Residues (PPR) 2015, EFSA Scientific Committee 2016) opens the door to considering landscape scales and multiple stressors in ERA.

In fact, landscape and spatially explicit ERA are specifically mentioned as an area for future development in the EFSA strategy for 2020 (see http://www.efsa.europa.eu/en/corporate/pub/strategy2020). Here is it stated that there will be a development of a holistic and integrated pan-EU approach in ERA with focus on a coordinated landscape-based framework across all relevant areas of EFSA’s remit (pesticides, feed additives, GMOs, plant health, animal health), and spatially explicit ecotoxicology and environmental fate and behaviour for pesticides.

3.3 Issues arising from landscape contexts for ERA

There are many issues that need to be addressed when considering a landscape context in European ERA for pesticides. The first and most obvious is that the landscape context needs to be defined. This is not an easy thing to do. Currently, the role of landscape context in landscape-scale ERA is not clear. Some few previous studies suggest that landscape structure might be a very important component (Topping and Odderskaer 2004, Dalkvist, Topping et al. 2009, Dalkvist, Sibly et al. 2013). It is also known that source-sink dynamics can lead to ‘action at a distance’ whereby the effect of pesticide application is seen at a location where no pesticide actually reached (Spromberg, John et al. 1998, Brock, Belgers et al. 2010, Topping, Craig et al. 2015). However, a broader coverage of organisms and ERAs is needed to determine how important factors such as these are.

A second important aspect is that the effect of landscape context may not be constant amongst species, and therefore species ecology and behaviour needs also to be considered. There are many potential factors here, including phenology, foraging behaviour, and reproductive output, all of which may alter the results of an ERA independently of the toxicological profile of the compound. For example, foraging behaviour that changes the likelihood of coming into contact with the pesticide as a result of active foraging preferences (e.g. for specific crops or crops in specific growth stages).

This project therefore seeks to answer the following questions for a landscape-scale population-level risk assessments:

- What is the role of landscape context when carrying out the ERA?
- To what extent do species ecology and behaviour influence the impact predicted in the ERA?

There are a number of other factors, which are not directly considered here. For example, how to cope with multiple stressors in a population-level assessment context. There is also a need to determine how to create standard scenarios for ERA on a European scale. Finally, if used as part of EU regulatory risk assessment then data access facilitating landscape assessments in all EU regulatory zones is needed. These factors are briefly considered in the discussion and recommendations for future work.

4. Methods Overview

This report includes extensive methods descriptions. Whilst important, these details are not necessary to follow the main approach and scenarios that were run for the main results of the project. These detailed methods also includes the details of the development and testing of the Great Crested Newt model that was developed during this project. This methods overview section therefore presents a short version of the methods necessary for the simulation results, whilst methods details and model-development is presented in a separated Detailed Methods section after the Discussion.

4.1 Approach

The approach used in this project was to consider population and landscape level risk assessment for a number of species either evaluating some factors/species-groups not currently considered by EFSA scientific opinions, or applying new concepts to species where this is novel. The aim was that the overall analysis should provide insight into identifying situations where current ERA practices may be under or over-protective of populations.

This section gives an overview of the approaches used and the strategy for model development and implementation that is extended in later sections.

The modelling approach used was an agent-based systems modelling approach using models already available or developed during this project in ALMaSS (Topping, Hansen et al. 2003) an agent-based model system. Using this approach, all individuals are modelled on daily time-steps, integrating mechanistic behaviour over a day, and representing the behaviour of individuals in a detailed representation of a landscape. The landscape represents the environment in which the population finds itself, and comprises weather, landscape structure, and the management carried out by man (primarily agricultural). The aim is that the animal models should react to this environment, and potentially each other, to create emergent population dynamics in space and time. These dynamics can then be compared between different scenarios, e.g. different pesticide usages.

In order to achieve this, the modelling aim is not to make the model as simple as possible, but to ensure that important details are not omitted, i.e. to avoid false exclusions (sensu Topping, Alroe et al. 2015). The idea is that the animal models should react to this environment, and potentially each other, to create emergent population dynamics in space and time. These dynamics can then be compared between different scenarios, in this case different pesticide application contexts, toxicities and species.

It is not possible to evaluate every possible combination of species ecology and landscape factors to determine which combinations may lead to problems in ERA. Initially, the intention was to create a generalised model that could be parameterised to represent any species by simulating combinations of parameterisable life-stages. This approach was attempted and models developed (see Appendix I), but was abandoned due to the difficulty of representing details necessary to realistically represent species ecology and behaviour critical to the outcome of the risk assessment. Therefore, the approach used here was to select a small number of cases that could be currently modelled within the time and resources available and were thought to cover cases identified as being of high risk. In addition, one species for which a model was not yet available was selected as a target for model development.

Selection of focal species therefore was based on those that were considered most at risk, and of most interest as focal species in ERA as it is currently carried out, and in one case, on the feasibility of model development. See the species selection section for details of the species chosen and the model representing them.

4.1.1 Landscape generation

A key part of this project was to be able to apply the model simulations to a range of landscapes from Denmark. In order to do this a largely automatic procedure was developed taking available GIS data, and farm information from central DK data sources and combining them to create highly detailed and accurate landscape maps for direct inclusions in ALMaSS scenario runs. The procedure for landscape generation is described in detail in the Detailed Methods section.

Using the landscape generation procedure, ten landscapes across Denmark were selected and 10 x 10 km sized maps generated (Table 1). See Detailed Methods for maps of each landscape.

Esbjerg (Es)		**Himmerland (Hi)**		**Karup (Ka)**		**Kolding (Ko)**		**Lolland (Lo)**	
Type	%Area	Type	%Area	Type	%Area	Type	%Area	Type	%Area
Conv. Cattle	75%	Conv. Cattle	65%	Conv. Potato	84%	Conv. Arable	59%	Conv. Beet	86%
Conv. Arable	10%	Conv. Pig	11%	Conv. Cattle	6%	Conv. Pig	20%	Conv. Arable	11%
Conv. Mixed Stock	4%	Conv. Mixed Stock	9%	Org. Cattle	3%	Conv. Cattle	12%	Conv. Mixed Stock	1%
Conv. Pig	4%	Conv. Arable	8%	Conv. Hobby	2%	Conv. Hobby	4%	Conv. Cattle	1%
Mors (Mo)		Næstved (Na)		Odder (Od)		Toftlund (To)		Tønder (Tn)	
Type	%Area	Type	%Area	Type	%Area	Type	%Area	Type	%Area
Conv. Pig	56%	Conv. Arable	61%	Conv. Pig	42%	Conv. Cattle	39%	Conv. Cattle	60%
Conv. Cattle	18%	Conv. Mixed Stock	16%	Conv. Cattle	14%	Org. Cattle	26%	Org. Cattle	17%
Conv. Arable	18%	Conv. Pig	8%	Conv. Arable	36%	Conv. Arable	14%	Conv. Arable	13%
Conv. Hobby	5%	Conv. Cattle	7%	Con. Hobby	4%	Conv. Pig	10%	Conv. Pig	5%

Table 1: Ten landscapes (Esbjerg – Tønder) with the four most common farm types by area listed. Conv. = conventional. Org. = organic. Abbreviations for the landscape names used throughout the report is in parenthesis after the landscape name.

4.2 Scenarios

The species used for simulation in the report were:

European brown hare (*Lepus europeaus*)
Skylark (*Alauda arvensis*)
A carabid beetle (*Bembidon lampros*)
Great Crested Newt (*Triturus cristatus*)

This selection of species represents mammals, birds and non-target arthropods, for which there is existing guidance on regulatory ERA, and amphibians for which there is no existing guidance, although EFSA is currently considering this. Existing models were available for the skylark, beetle and hare, but the Great Crested Newt model was developed within this project.

4.2.1 Common scenario details

Unless specifically stated otherwise the following setting were used for all simulations:

- All scenarios were run on one of the 10 x 10 landscapes described above. Thus, the effect of scale of assessment was not considered, but scale of use within the landscape was part of the standard scenarios.
- Simulation lengths were 30 years.
- Weather used was from central Jutland from 1984-2014 from the E-OBS dataset (Haylock, Hofstra et al. 2008).
- Pesticide impacts were calculated by comparison to an identical scenario with the exception that the specific pesticide under testing was not applied (and nothing was applied to replace it).
- When spraying focal pesticides (i.e. those where the impact was evaluated), drift was included, assuming that drift occured up to 12 m from the edge of any sprayed field, following the equation $p = e^{-0.6122x}$, where p is the proportion of application rate falling from x to 1m, and x is distance in m from the point of spray. This gives ca. 24% drift at 1 m, and 2.1% at 5 m. The direction of drift varies randomly depending on the day of spraying, and was assumed to be due north, south, east or west.
- Once applied a pesticide was assumed to have a standard environmental half-life of 7 days, which means that after 7 days, the concentration of pesticide in any location would be 50% of the applied rate (assuming a single application). This is achieved by removing a fixed proportion of pesticide per day. Subsequent applications add to the environmental concentration already present, and further decay considers the summed concentration.
- For simplicity, any animal assumed to be feeding in a contaminated location or otherwise exposed received the full amount of environmentally available pesticide. This is a method to reduce the number of variables needed to be considered in the comparisons, but means that exposure is higher than would be realistic.

Due to the large number of scenarios and replicates and the long run times (e.g. 48 hours for beetle simulations), the simulations were run on a powerful Linux cluster Grendel-slurm (see http://phys.au.dk/forskning/cscaa). Grendel-slurm consists of 88 nodes. Each node has two Intel Xeon E5-2680 v4 CPU's @ 2.4 GHz (providing a total core count of 28 pr. node), 256 GB memory, 6TB local disk.

4.2.2 Standard scenarios

To compare impacts between species, a standard set of toxicity, exposure and landscape scenarios was created and applied to all species, representing a fixed set of toxicity and exposure conditions. These scenarios represented a range of landscape conditions, including the farming that was carried out on those landscapes, and a number of combinations of toxicity and area and timing of application scenarios. This provides a standard set of scenarios within which the impact of the pesticide application can be compared. However, this comparison is made difficult because of differential sensitivities and assumed effects on different species. For example, direct mortality of vertebrates should not occur, hence the only useful mode of action considered is via reproduction or growth, in contrast non-target arthropods might be in the field when an insecticide is applied and thus might be expected to experience acute mortality. Once a pesticide has been applied in the model it undergoes an environmental degradation.

To further help to reduce the number of variables needed to be considered, the toxicity of all pesticides applied was assumed to be a multiple of the dose that would elicit and effect in an individual. The application rates used were X1, X10, X100 and X1000 the dose that would trigger an effect. This means that any individual in a field at the time of spraying an X1 dose would exhibit an effect on that day, but an individual entering the field on the following day would not exhibit an effect due to environmental degradation. This method means that we need only consider the application rate when comparing between species, since a X10 increase in rate will mean a proportional X10 increase in exposure.

At the same time, the expression of toxicity was assumed to be specific to the species under consideration, e.g. beetles would exhibit direct mortality as a result of exposure whilst hares might suffer reproductive depression. This permitted direct comparisons between species whilst still retaining their specific responses. The aim was thus to be able to compare species specific responses to changes in toxicity, exposure, and landscape factors to determine which combinations result in population vulnerability.

Pesticide applications differed between the standard crops (although in the cereals these percentages are increased compared to typical Danish conditions). Unless specified otherwise due to specific scenario modification the pesticide probability of application and times are listed in Table 2.

Crop	Application 1	Application 2	Application 3
Winter Wheat	50% on 15th May	50% on 1st June	50% on 14th June
Spring Barley	35% on 15th May	35% on 1st June	35% on 14th June
Oil Seed Rape*	21% on 15th April	7% on 1st May	5% on 15th May

Table 2: The timing of the day of application for each of three pesticide applications for the three crops used in ERA scenarios.*OSR applications were dependent upon previous applications, i.e. 7% is 30% of the 21% that applied the first application.

Eleven standard scenarios were run for each species and for each of the ten landscapes created for the study. The factors that were varied were the application rate, the crops to which pesticide was applied (at X 10 application rate), and whether a spring barley monoculture or normal cropping was assumed. For spring-barley monoculture the probability of pesticide application was altered from 35% (Table 2) to 100%. The factors varied for the standard scenarios are summarised in Table 3.

Reference Name	App. Rate Winter Wheat	App. Rate Spring Barley	App. Rate Oil Seed Rape	Crop Distribution
Standard	0	0	0	Normal
SB100%	0	0	0	Spring Barley
WSOx1	1	1	1	Normal
WSOx10	10	10	10	Normal
WSOx100	100	100	100	Normal
WSOx1000	1000	1000	1000	Normal
WSX10	10	10	0	Normal
Sx10	0	10	0	Normal
Wx10	10	0	0	Normal
Ox10	0	0	10	Normal
SBX10	NA	10	NA	Spring Barley
SB100X10	NA	100% of fields X10	NA	Spring Barley

Table 3: The eleven standard scenarios and the factors varied for each. Scenarios are referenced by their reference name in the rest of the report.

For each basic crop scenario the proportion of the landscape treated varied depending on the landscape and the farming it contained. WSO scenarios from 39% to 68% with the highest proportion of fields treated, to the O scenarios with 1.6% to 8.8% of fields treated (Table 4).

Scenario	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSO	0.393	0.477	0.365	0.675	0.533	0.659	0.650	0.683	0.439	0.436
WS	0.362	0.435	0.349	0.589	0.512	0.575	0.568	0.595	0.403	0.402
W	0.110	0.145	0.069	0.276	0.230	0.275	0.260	0.284	0.128	0.123
S	0.252	0.290	0.28	0.313	0.282	0.300	0.308	0.311	0.275	0.279
O	0.031	0.042	0.016	0.086	0.021	0.084	0.082	0.088	0.036	0.034

Table 4: The proportion of all agricultural fields treated with the focal pesticide in each landscape for each crop scenario

4.2.3 Species-specific ERA scenarios

In addition to the standard scenarios, specific scenarios were developed for special cases where it was considered that a particular facet of the system was of importance to the outcome of the ERA.

4.2.3.1 Hare

The hare was used to investigate the interaction between landscape and farming. The question being to what extent is the result of an ERA reliant on landscape structure, and to what extent it is due to the farming that is carried out. Five landscapes were used for the analysis: Esbjerg, Karup, Lolland, Odder and Tønder, representing the range of density options from the baselines. The pesticide was applied to winter wheat and spring barley at X5 application rate to generate a medium level response. Each of the five landscapes used were run with five different farming conditions, i.e. those originating from all five landscapes. Thirty-year simulations were run with 20 replicates for each combination of landscape and farming, i.e. 25 combinations.

4.2.3.2 Beetle

The beetle model was used to illustrate the effect of assumptions that ERA can be applied to different stages of the life-cycle independently. In this case the beetle is assumed to be covered by ‘non-target terrestrial invertebrate ERA’ but its larvae are soil dwelling and are therefore treated separately under ‘in soil organisms’ ERA.

Here we evaluated the impact of the same scenario on larvae, adults separately and together. Scenarios were run assuming X1 application rate to winter wheat, spring barley and oil seed rape and impacts on beetle larvae and adults as acute mortality probability if the local environmental concentration of pesticide was above a trigger concentration. This concentration was initially assumed to be the same for both life-stages to allow direct comparison of effects, but larval impacts were negligible, hence the scenarios were re-run with adults using X1 application rate and normal sensitivity, and larvae using X1 application rate and X10 sensitivity (AX10 & LX10). A third scenario with X1 application rate but combining both AX10 & LX10 effects was also run (ALX10).

4.2.3.3 Skylark

The skylark is generally well-protected since under current ERA guidance for birds and mammals direct reproductive impacts ought to be observed in reproduction tests carried out on mallards or quails (EFSA 2009). One possible effect that might be missed, however, is the impact of eggshell thinning. This is a well-known potential effect of pesticides, particularly well-known for organochlorine pesticides such as DDT & DDD (Hickey and Anderson 1968, Anderson and

Hickey 1970, Cooke 1973), and should be evaluated in current ERA (EFSA 2009). However, eggshell thinning itself does not cause a problem, it is only a problem if the egg subsequently cracks. Therefore, there is a question as to the biological relevance of this issue. Here we showed how this might be evaluated by making the assumption that exposure to a pesticide leads to eggshell thinning to the point where the egg will crack and implementing the following scenarios to evaluate the population outcome under different assumptions.

Eggshell Scenario 1 - Using the standard crop scenario (based on real crops present in 10 landscapes), but assuming global exposure (i.e. all birds were assumed to forage from pesticide treated fields). Pesticide effect was assumed to be eggshell reduction resulting in clutch loss at 5 to 75% chance. The impact relative to the baseline population size was determined for all 10 standard Danish landscapes. This forms a worst case scenario from which the clutch loss effect can be evaluated.

Eggshell Scenario 2 - In this scenario all winter wheat, spring barley and oilseed rape crops received three applications of pesticide at the standard rate (120% of concentration required to trigger effects) at 14 days apart. Although the pesticide-induced effect was different, this scenario was identical to the standard WSOX1 scenario.

a) If exposed above the threshold 50% or 75% clutch loss was assumed.
b) As a second set of assumptions the effect of using individual egg loss (i.e. the 50 or 75% chance is applied to all eggs individually rather than per clutch).

Eggshell Scenario 3 - This scenario was identical to scenario 2 except that a monoculture of spring barley was used rather than normal crop proportions. First application of pesticide in mid-May following standard spring barley application schedule. Assumptions of 75% chance of effect if exposed during laying above the trigger value and X1 and X1000 application rate. This scenario was only applied to Esbjerg landscape.

a) If exposed above the toxicity threshold 75% clutch loss was assumed.
b) If exposed above the toxicity threshold 75% egg loss was assumed.

Eggshell Scenario 4 - As for Scenario 3, but the timing of the spray was advanced by 28 days. Hence, this scenario considered a monoculture of spring barley and standard X1 and X1000 application rates but with an earlier application of the pesticide than standard. This scenario was only applied to Esbjerg landscape.

a) If exposed above the toxicity threshold 75% clutch loss was assumed.
b) If exposed above the toxicity threshold 75% egg loss was assumed.

For all scenarios, to save processing time, all egg-shell test scenarios were run for a minimum of 10 replicates, and then further replicates were run until the 95% confidence limit of the final 10-year population mean was within 2% of that population mean. This was achieved between 10 and 20 replicates in all cases.

In addition to the above scenarios, an example of assumption of total clutch loss for all birds after 10 years simulation provided a background against which the scenarios could be evaluated.

4.2.3.4 Great Crested Newt

The newt was used to exemplify four points about population ERA and landscape scale. The first three were based on a set of overspray scenarios. These scenarios were identical to the WSO standard scenarios except that the impact was assumed to be overspray only, hence only free-living newts directly in contact with the sprayed pesticide were affected. In all cases, we assumed a lethal effect concentration (LCx) and applied this probability of mortality for the single day of application only. The factors evaluated were:

- Impact of varying LCx on population size and AOR-scores. LC_0 to LC_{100} in steps of 5 were applied to the population in the Næstved landscape using the latest weather series available 1984-2014 (10 replicates).

- The effect of changing weather inputs. LC_0 to LC_{100} in steps of 10 were applied to the Næstved landscape using weather data from 1950-1979 (5 replicates).
- The length of time taken for recovery at population levels was evaluated on Mors and Næstved landscape using the LC100 scenario and application of pesticide for 1- and 10-years during a 30-year simulation run (5 replicates).

The fourth point investigated was the implications of using stochastic pond quality rather than global optimal pond quality. This provides insight into the effect of including habitat quality variability on the newt ERA, but using a stochastic rather than mechanistic implementation. This was investigated by running all standard scenarios with 40 replicates (and 80 baseline replicates), as well as the LCx scenarios on Næstved (120 replicates of each). Data is presented for the major effects compared to global optimal ponds, and consideration of the importance of the between run variation.

4.3 Measurement Endpoints Used

Population descriptors used as measurement endpoints in the ALMaSS simulations need to fulfil some basic criteria. They should:

- Show changes in the density of animals present in the landscape as a result of pesticide application
- Show changes in the distribution of animals in the landscape as a result of pesticide application
- Show long-term impacts of continued use (i.e. changes in abundance and distribution with time)
- Should be easily automated and provide a readily assessable metric

Measures typically used in population ecology are counts of numbers per unit area (density) and changes in density with time as a time-series. One commonly used method to do this in an risk assessment process is Population Viability Analysis (PVA). PVA traditionally defined as the process that determines the probability that a population will go extinct within a given number of years. It is a statisically based approach, individually developed for a target population or species. Each PVA is therefore unique and has the aim of ensuring that a population is viable over the long-term (Sanderson 2006). PVA is however a complex process and not suited to automated assessment as is required here.

One possibility is simply to work with the population density with time for fixed location. However, this simple metric has drawbacks when comparing across different species with different densities under the same conditions, and across different landscapes which may have different baseline (untreated with pesticide) populations. A solution is to combine the baselines and treatment scenarios results and make a time-series relative to the baseline by dividing the difference between treatment and baseline by baseline (Figure 1). This has the advantage that different densities can be compared on a relative scale. However, the measurement of effect is difficult since impacts change with time, typically increasing with time, but at a diminishing rate of increase (Figure 1). Similarly, if recovery is needed to be measured the point at which it is measured will affect the value chosen.

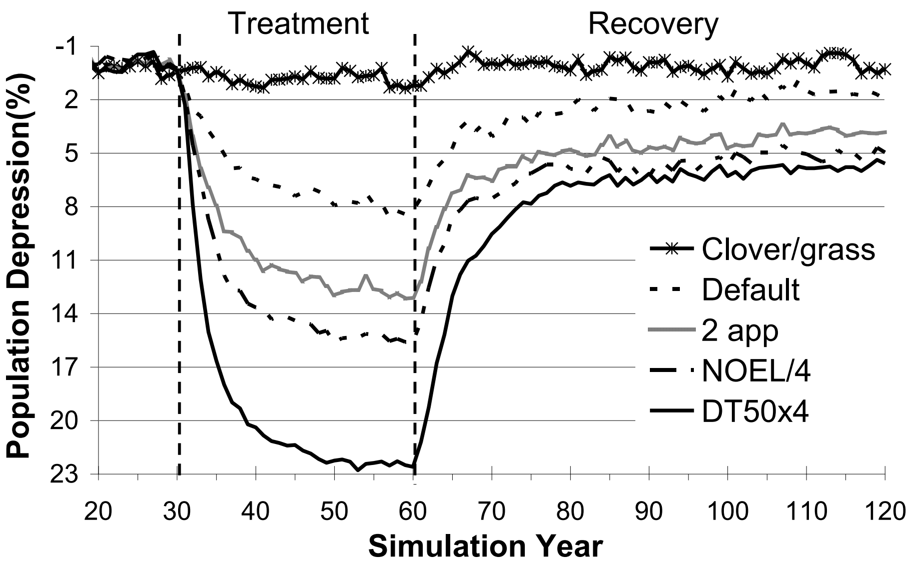


Figure 1: Population depression calculated by taking the difference between treatment and baseline as a percentage difference with time. Unity means no difference from the baseline calculated as (1- ((Treatment-Baseline)/Baseline). In this example pesticides were applied between the two vertical dotted lines from (Dalkvist, Topping et al. 2009).

4.3.1 AOR-Index

To overcome the spatial limitations the AOR-Index was developed by the ALMaSS group (Hoye, Skov et al. 2012). This is a simple statistic used to describe the impact of changes in drivers in a simulation relative to a reference (baseline) condition.

The index is based on one of the most important large-scale biodiversity patterns i.e. the generally positive relationship between local population density and site occupancy, the abundance–occupancy relationship (AOR). A distinction is made between the interspecific AORs and the intraspecific AORs, where the former describes the relationship forming among multiple species and the latter describes changes in abundance and occupancy through time. Scenarios of population development for single species can thus be quantified by the intraspecific AOR relating current to future conditions. Positive AORs have been documented across a broad range of taxa in many habitats (see Gaston et al. 2000 for a review).

The aim of developing the AOR-index was to provide a tool to interpret changes in both abundance and distribution across species to changes in landscape structure and composition as well as the abiotic environment. The AOR-index is derived from locations of individuals of a particular model species. To create it, the landscape is overlaid with a regular grid and the number of animals in each grid cell was recorded. Occupancy (*O*) is quantified as the proportion of grid cells occupied by at least one animal and abundance (A) as the mean number of animals in occupied grid cells. The following relationship follows by the definitions:

$$A \equiv \frac{I}{(1-O)\cdot G}\ ,\ AOR = \frac{I}{(O-O^2)\cdot G} \qquad (1)$$

where *I* is the total number of individuals in the simulated area and *G* is the total number of grid cells. The set (A, I, O) is characteristic for the response and if two values are known then the third value can be calculated using the definition in Eq. 1. Typically *O*>0 for smaller grid size, where some grids are not occupied, while for larger grid sizes the value of *O* tend to approach zero, depending on the spatial distribution of the individuals. Different species need different sized grids to account for the differences in the spatial scale at which the species use the landscape. If the grid cell size is too small, abundance will approach one individual per grid cell and if the grid cell size is too large, all cells will be occupied and the AOR-index will have limited sensitivity to detect a change in abundance or occupancy with too small or too large grid cell sizes, respectively. The optimal grid cell size depends on the distribution of individuals of the particular species. Three heuristic rules are used to identify the optimal grid cell size for a model species. Firstly, the grid cell size should lead to at least 250 grid cells in total. Secondly, all grid cell sizes where more than 50% of the cells were occupied are excluded. If this excludes all grid cell sizes then a minimum grid cell size is chosen fitting with the 1st and third rules. Thirdly, the mean density of individuals in all occupied grid cells is counted with a grid cell size with a mean density closest to five individuals preferred. The first rule was introduced to maintain resolution in the measure of occupancy to allow small distributional changes to be detected. The second rule ensured that both positive and negative changes in occupancy would be detected and the third rule ensured some variation in abundance. Without the last rule, abundance for territorial species would approach one female per grid cell and be insensitive to change. (Hoye, Skov et al. 2012) created AOR-Indices for six ALMaSS species. The resulting grid cell sizes were 50 m for the beetle and the spider, 100 m for the vole, 200 m for the skylark, and 400 metres for the hare and the partridge. In this report these grid sizes are used, and a 400m grid size used for the newt model, which was not part of the original publication.

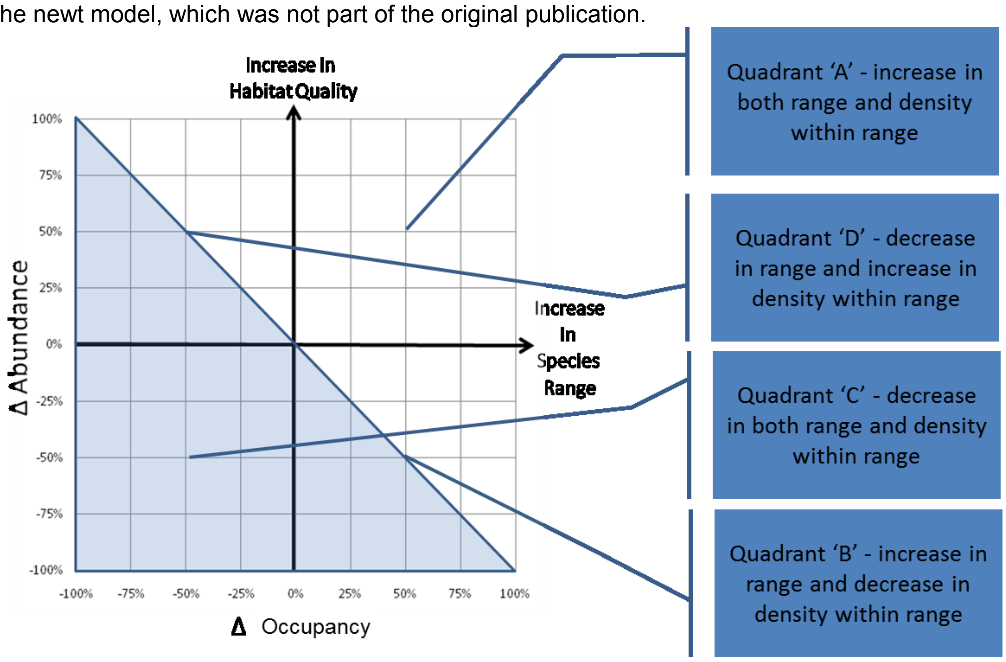


Figure 2: Properties of the AOR-Index assuming its use as a relative measure.

When AOR-Indices are plotted as a relative measure compared to a baseline condition then spatial distribution, or change in spatial distribution between treatment and baseline scenarios forms a part of the metric (Figure 2). This method provides a quick visual check of the impact of a particular management.

When comparing between scenarios it is possible to quickly assess the relative impact of different treatments compared to the same baseline. For example, the impact of the same pesticide in different landscape types can be compared, here using the ALMaSS hare model (Figure 3). It is easy to visually interpret the graphs and identify those landscapes where the use of the pesticide would cause most concern. Here all landscapes can be directly compared since the scenario is the same, even though the baseline populations were different.

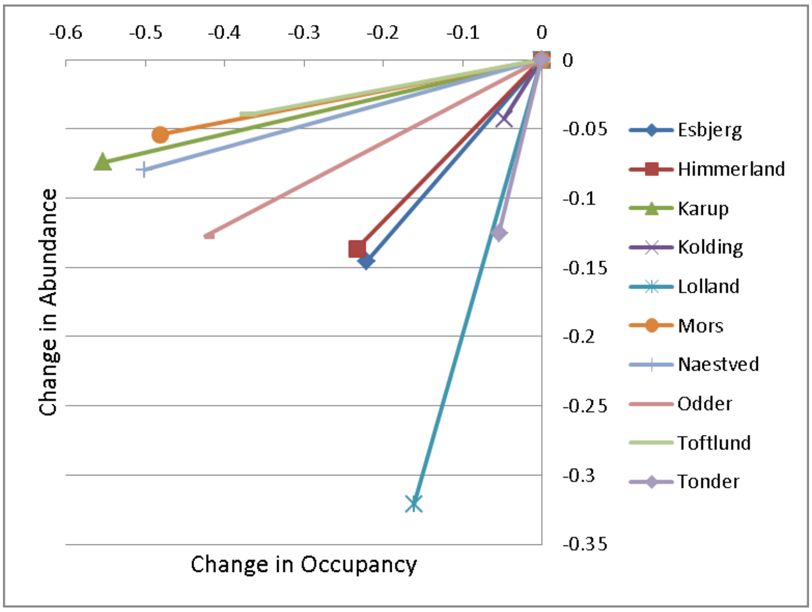


Figure 3: AOR-plot for an ERA for an endocrine disruptor impact on brown hares in 10 different Danish landscapes. The length of the line indicates the relative of the impact compared to the baseline condition for each landscape.

4.3.2 Other endpoints

In addition to using AOR-Index, two other endpoints are used. The first is a simple measure of change in population size relative to the baseline (Figure 1). This has the advantage that it removes the variation caused by different landscape and allows direct comparison with the baseline. Since for the most part recovery was not of interest, only the treatment period was assessed.

The other metric is a measure of population growth rate (pgr). This was obtained by dividing a time series of relative population size up into ten-year sections, then using linear regression to determine the slope of the section plotting relative population size in the treatment against time. This differs from the measure of impact in that the slope may be zero even if the population is radically reduced. If so, this indicates long-term population stability, if less than zero it indicates population decline. This is therefore not pgr in the typical ecological sense, but a relative pgr designed to be used to compare two dynamic scenario outputs.

5. Results

The results section is divided up initially by species. Here all results of all scenarios for that species are presented starting with the standard scenarios. There is then a short analysis of the combined set of species/scenarios tested under standard scenarios.

5.1 European brown hare ERA

5.1.1 Standard scenarios

Hare density in the baseline scenarios varied from 0.95 to 15.79 females km^{-2} depending upon the landscape considered. Highest densities were found in the sugar beet landscape and lowest in the landscape dominated by potatoes. High densities were also found in Tønder landscape, which has a high proportion of permanent grass (Table 5).

Baseline	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Standard	5.68	5.22	0.95	6.11	15.79	1.49	2.44	3.47	0.95	8.09
SB100%	5.70	4.99	1.00	6.10	15.26	1.60	2.60	3.76	0.92	7.98

Table 5: The mean landscape density of female hare km^{-2} in the last 10 years of simulation (years 21-20) predicted for the standard baseline and 100% spring barley baseline scenario for all landscapes (Es-Tn).

Population sizes relative to the baseline for all standard scenarios are collected in Table 6. Increasing toxicity from X1 to X1000 increased the impact of the pesticide, with X1000 having 1.4 to 4.6 times the impact of the X10 scenario, and the X1 scenario having no population impact. Relative increase in impact was greatest for Næstved and least for Tønder (Figure 4 & Table 7. Impact was correlated loosely with area treated for X10 toxicity scenarios (Figure 5), but this did not fully explain the between landscape variability. There was no significant relationship between population size and impact (p=0.69, df 9). In all cases of X10 application rate, maximum impacts were found in the SB100X10 scenario, the range relative to baseline being 0.45 to 0.63. There was no correlation between the WSOX10 scenario impacts and the SB100X10 impacts (R^2 0.0049).

Scenario	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	1.00	0.96	1.04	1.00	0.97	1.00	1.06	1.09	0.96	0.99
WSOx10	0.67	0.64	0.70	0.67	0.57	0.52	0.54	0.59	0.60	0.85
WSOx100	0.49	0.57	0.62	0.47	0.45	0.34	0.29	0.37	0.60	0.77
WSOx1000	0.29	0.36	0.36	0.26	0.28	0.14	0.12	0.19	0.29	0.62
WSX10	0.66	0.68	0.75	0.65	0.59	0.52	0.54	0.58	0.56	0.86
Sx10	0.74	0.77	0.77	0.75	0.66	0.66	0.71	0.72	0.68	0.89
Wx10	0.74	0.73	0.86	0.70	0.65	0.63	0.62	0.70	0.65	0.89
Ox10	0.79	0.78	0.86	0.87	0.79	0.72	0.75	0.84	0.87	0.91
SBX10	0.80	0.75	0.77	0.71	0.77	0.70	0.86	0.88	0.71	0.80
SB100X10	0.49	0.45	0.51	0.53	0.63	0.45	0.53	0.59	0.51	0.56

Table 6: Mean female hare population size relative to baseline for the last ten years of simulation (years 11-20 of pesticide application) for all standard scenarios and landscapes (Es-Tn).

	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Initial Effect	0.003	-0.044	0.045	-0.002	-0.033	0.005	0.063	0.085	-0.036	-0.013
Slope	-0.231	-0.186	-0.212	-0.241	-0.216	-0.276	-0.309	-0.289	-0.201	-0.119

Table 7: The initial impact of X1 toxicity exposure and the rate of change in population depression per X10 increase in toxicity for all 10 landscapes assuming application to winter wheat, spring barley and oilseed rape for hares. Slope fitted using linear estimation over 1X to1000X.

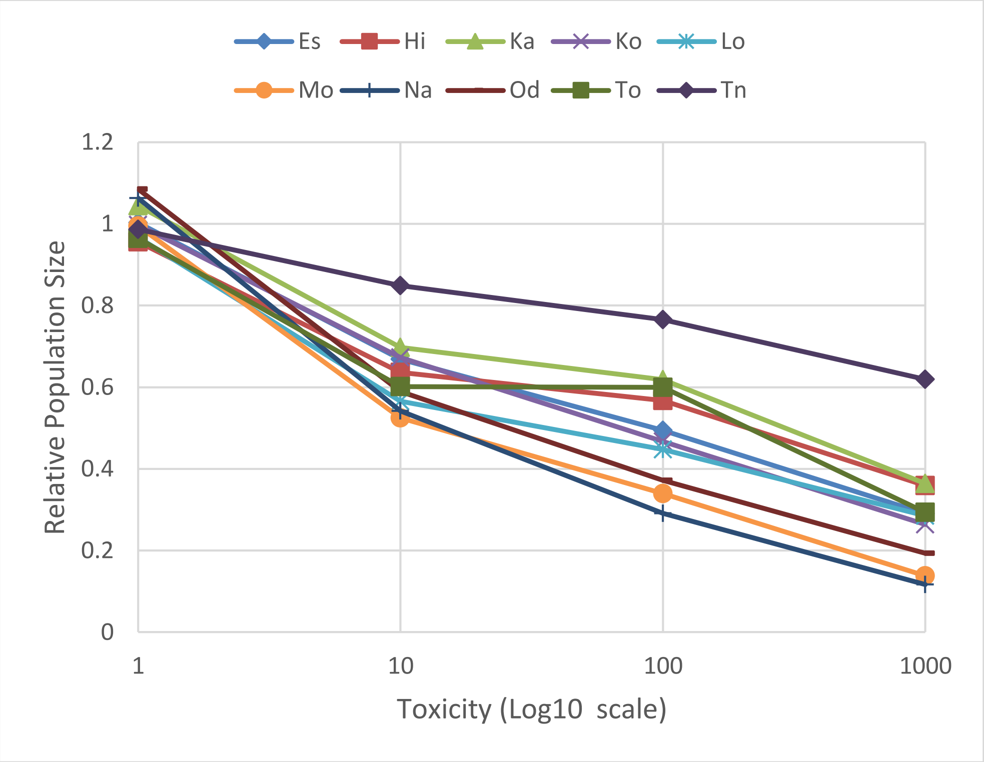


Figure 4: Changing impact of pesticides on hares with increasing toxicity by factors of 10 for all ten standard landscapes (Es-Tn) assuming application to winter wheat, spring barley and winter oil seed rape.

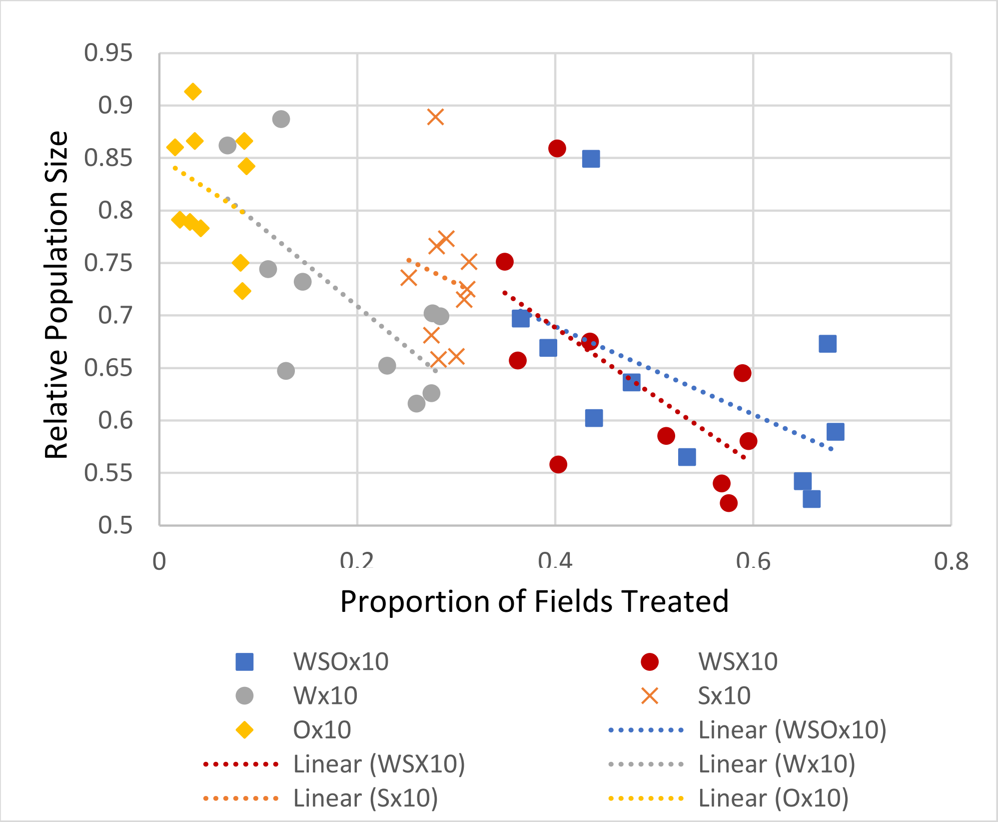


Figure 5: Relative hare population size plotted against area treated for WSOX10, WSX10, WX10, SX10, & OX10 scenarios with fitted regression lines.

Analysis of relative population growth rate indicated that in the majority of scenarios with toxicity X10 or higher the population was still decreasing after 10 years of application. Tønder and Lolland, the landscapes with the highest density of hares, had the lowest number of declining populations in the second 10 years of application. This indicates that although impacts in terms of relative population size may be low (e.g. 0.84 in Karup OX10), the population was still in decline relative to the baseline conditions.

Scenario	**Es**	**Hi**	**Ka**	**Ko**	**Lo**	**Mo**	**Na**	**Od**	**To**	**Tn**
WSOx1		-0.002								
WSOx10	-0.011	-0.013	-0.010	-0.010	-0.003	-0.019	-0.022	-0.020	-0.010	
WSOx100	-0.013	-0.013	-0.013	-0.010		-0.019	-0.018	-0.016	-0.020	-0.004
WSOx1000	-0.013	-0.015	-0.021	-0.012	-0.005	-0.015	-0.015	-0.018	-0.020	
WSX10	-0.011	-0.012		-0.012	-0.004	-0.024	-0.018	-0.017	-0.016	
Sx10	-0.016	-0.005	-0.023	-0.006		-0.013	-0.011	-0.008	-0.008	
Wx10	-0.012	-0.015	-0.012	-0.004		-0.016	-0.009	-0.019	-0.008	
Ox10		-0.006	-0.006		-0.003	-0.020	-0.005	-0.012		
SBX10	-0.019	-0.011	-0.008			-0.025		-0.006	-0.027	-0.015
SB100X10	-0.031	-0.019	-0.010	-0.012		-0.018	-0.014	-0.018	-0.019	-0.015

Table 8: The fitted regression slopes for hare population depressions relative to baseline for all standard scenarios and landscapes for the second 10 years of application where the slope was significant at p<0.05.

AOR index scores show a similar pattern of impact to the relative population sizes, however, here it is clear that impacts on the range of the hares and their densities were very much landscape and scenario dependent. For example in the Lolland landscape impacts were primarily on abundance, and occupancy was maximally reduced by 36% for the SB100X10 scenario, whereas in Karup abundance was maximally reduced by 8%, but occupancy by up to 61%. In the two monoculture-scenarios SBX10 and SB100X10, impacts were primarily on occupancy with abundance affected only by 1-10%, whereas other cropping scenarios had mixed effects across landscapes.

Scenario	A/O	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	A	0.00	-0.02	0.01	0.00	-0.03	0.00	0.02	0.03	0.00	-0.01
	O	0.00	-0.02	0.04	0.00	0.00	0.00	0.05	0.05	-0.03	0.00
WSOx10	A	-0.18	-0.18	-0.03	-0.19	-0.38	-0.07	-0.12	-0.15	-0.04	-0.12
	O	-0.18	-0.22	-0.28	-0.17	-0.08	-0.44	-0.39	-0.30	-0.37	-0.04
WSOx100	A	-0.27	-0.22	-0.03	-0.30	-0.48	-0.09	-0.17	-0.23	-0.03	-0.18
	O	-0.33	-0.28	-0.36	-0.33	-0.14	-0.63	-0.65	-0.52	-0.38	-0.07
WSO x1000	A	-0.36	-0.30	-0.06	-0.39	-0.60	-0.12	-0.21	-0.27	-0.06	-0.29
	O	-0.55	-0.48	-0.61	-0.57	-0.28	-0.84	-0.85	-0.73	-0.69	-0.13
WSX10	A	-0.19	-0.17	-0.03	-0.21	-0.37	-0.07	-0.12	-0.15	-0.04	-0.11
	O	-0.19	-0.19	-0.23	-0.19	-0.08	-0.44	-0.39	-0.31	-0.42	-0.04
Sx10	A	-0.15	-0.12	-0.02	-0.15	-0.30	-0.05	-0.08	-0.11	-0.02	-0.08
	O	-0.14	-0.12	-0.22	-0.12	-0.05	-0.31	-0.22	-0.19	-0.30	-0.03
Wx10	A	-0.14	-0.14	-0.01	-0.17	-0.31	-0.06	-0.10	-0.12	-0.03	-0.08
	O	-0.13	-0.15	-0.13	-0.15	-0.06	-0.34	-0.31	-0.21	-0.33	-0.03
Ox10	A	-0.12	-0.11	-0.01	-0.08	-0.19	-0.04	-0.06	-0.06	-0.01	-0.07
	O	-0.10	-0.12	-0.13	-0.06	-0.03	-0.25	-0.20	-0.10	-0.13	-0.02
SBX10	A	-0.03	-0.04	-0.04	-0.05	-0.01	-0.02	-0.01	-0.01	-0.01	-0.03
	O	-0.17	-0.22	-0.20	-0.25	-0.22	-0.29	-0.14	-0.11	-0.28	-0.18
SB100 X10	A	-0.08	-0.10	-0.08	-0.08	-0.01	-0.04	-0.02	-0.04	-0.02	-0.07
	O	-0.46	-0.50	-0.45	-0.43	-0.36	-0.53	-0.46	-0.38	-0.48	-0.40

Table 9: AOR-index scores for hares under all standard scenarios and landscapes (Es-Tn, A = abundance, O = occupancy).

5.1.2 Contribution of landscape and farming to ERA impacts

Each of the five landscapes used were run with five different farming conditions, i.e. those originating from all five landscapes. The result was a widely different baseline from 0.1 to 15.1 adult hares km^{-2} (Table 10). This is compared to the range of densities for the landscapes with their respective farming of 0.9 to 1.2 females km^{-2}. The mean range of population densities when varying landscape over farming was 6.16, whereas the range of population densities when varying farming over landscapes was 11.6, approximately double. Therefore we can conclude that at least for this set of landscape and farming, the effect of variation in farming was approximately twice that of variation in landscape.

		Farming					
		Es	**Ka**	**Lo**	**Od**	**Tn**	**Range**
Landscape	**Es**	5.1	2.6	15.1	6.1	5.1	**12.5**
	Ka	3.5	0.9	14.8	3.8	3.6	**13.9**
	Lo	0.3	0.1	12	0.3	0.2	**11.9**
	Od	2.5	1.1	14.3	3.1	2.2	**13.2**
	Tn	7.5	6.1	12.6	7.5	7.5	**6.5**
	Range	**7.2**	**6.0**	**3.1**	**7.2**	**7.3**	

Table 10: Mean adult female numbers km^{-2} for five landscape and farming combinations under baseline conditions. Range is the difference between the maximum and minimum value for rows and columns respectively.

Using the same scenario but in all combinations of five landscapes and their five different farming conditions produced a wide range of impacts relative to the baseline conditions above. Population impacts were from the lowest, Lolland landscape and Tønder farming, at 0% to the highest, Lolland landscape with Esbjerg farming at 29%.

AOR-index scores are shown for each combination as a both landscape and farming oriented plots (Figure 6-Figure 10). Different landscape farming appear to have different effects depending upon the combination. For example, Lolland landscape has medium to large occupancy impacts in 4 out of 5 case, the exception being Lolland farming where the only large change in in abundance. Lolland farming results in rather small pesticide impacts, and in all cases changes are in abundance rather than occupancy.

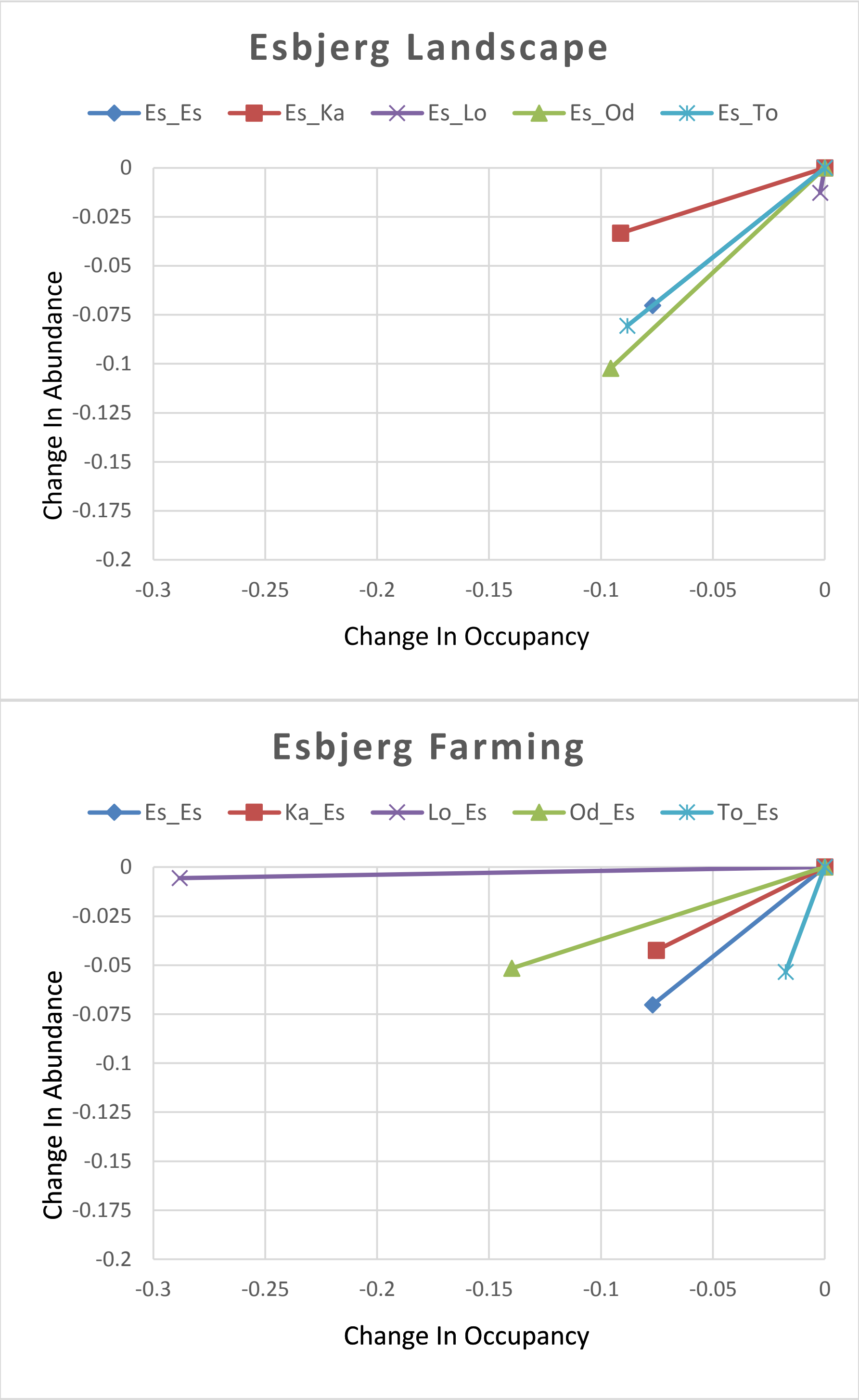
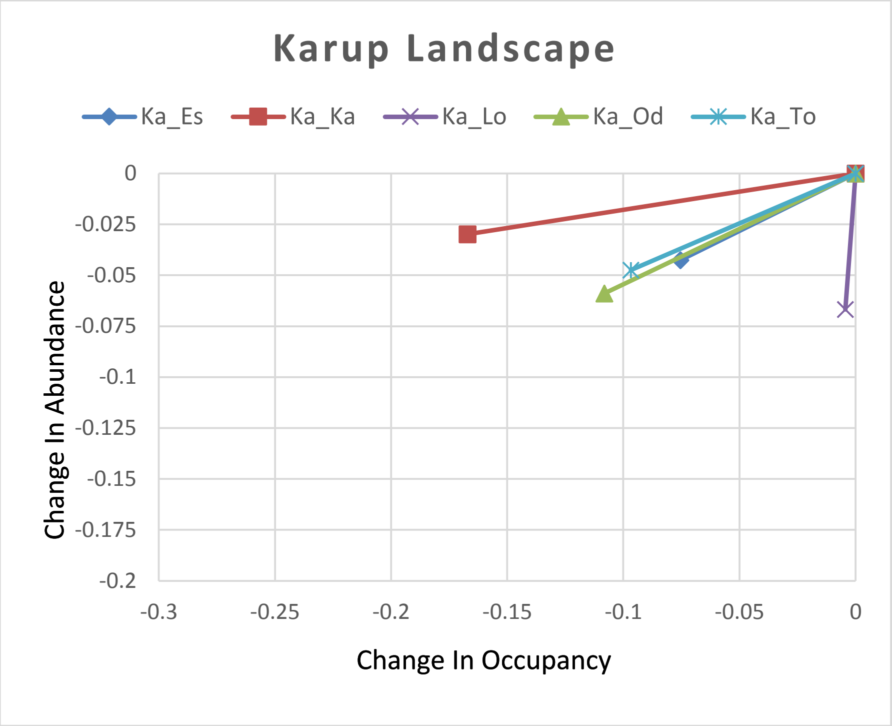


Figure 6: AOR-plots for hare ERA on Esbjerg landscape with Esbjerg, Karup, Lolland, Odder and Tønder farming (above) and Esbjerg farming on Esbjerg, Karup, Lolland, Odder and Tønder landscapes (below).



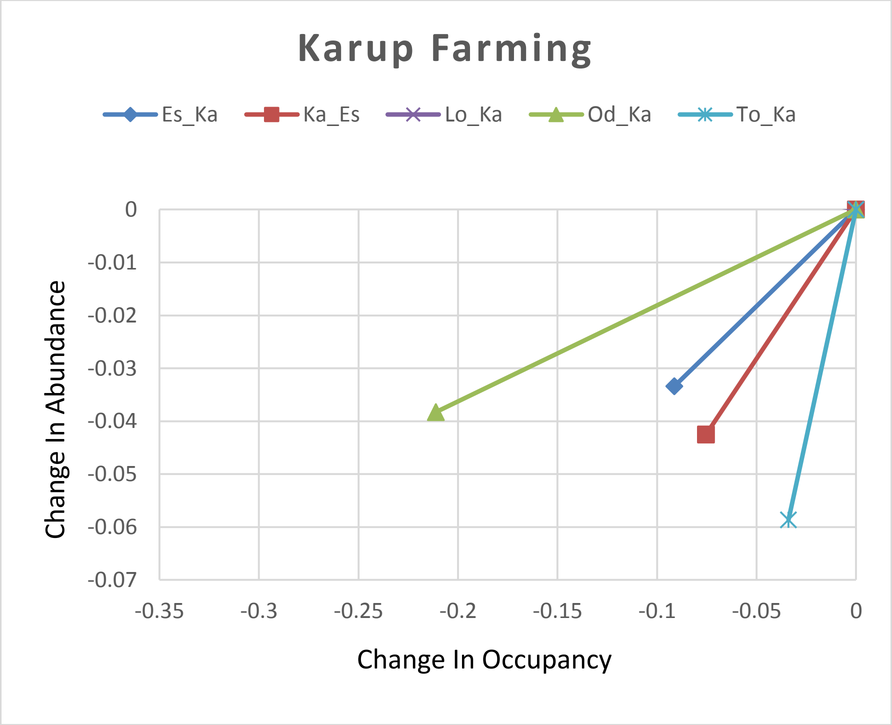


Figure 7: AOR-plots for hare ERA on Karup landscape with Esbjerg, Karup, Lolland, Odder and Tønder farming (above) and Karup farming on Esbjerg, Karup, Lolland, Odder and Tønder landscapes (below).

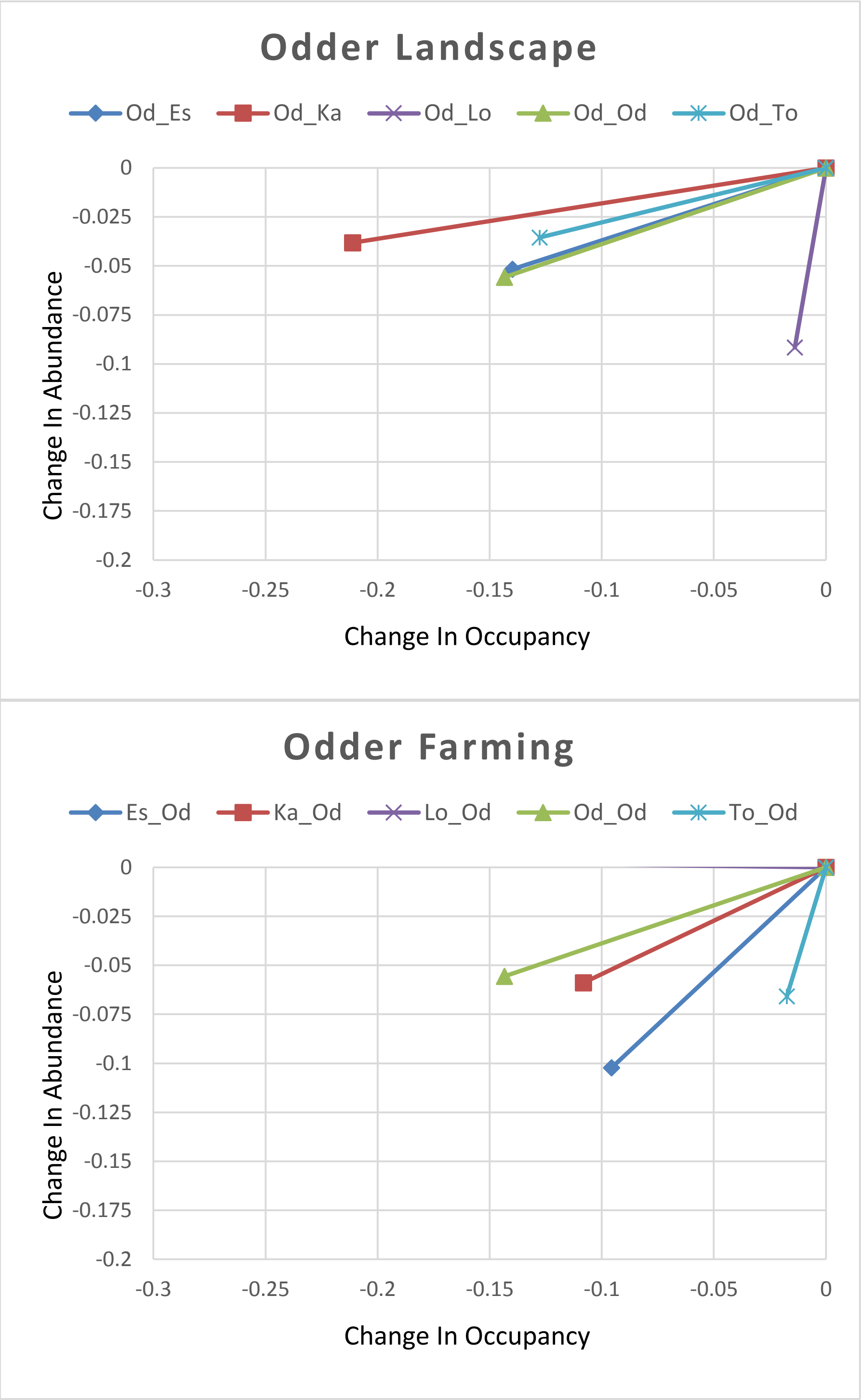


Figure 8: AOR-plots for hare ERA on Odder landscape with Esbjerg, Karup, Lolland, Odder and Tønder farming (above) and Odder farming on Esbjerg, Karup, Lolland, Odder and Tønder landscapes (below).

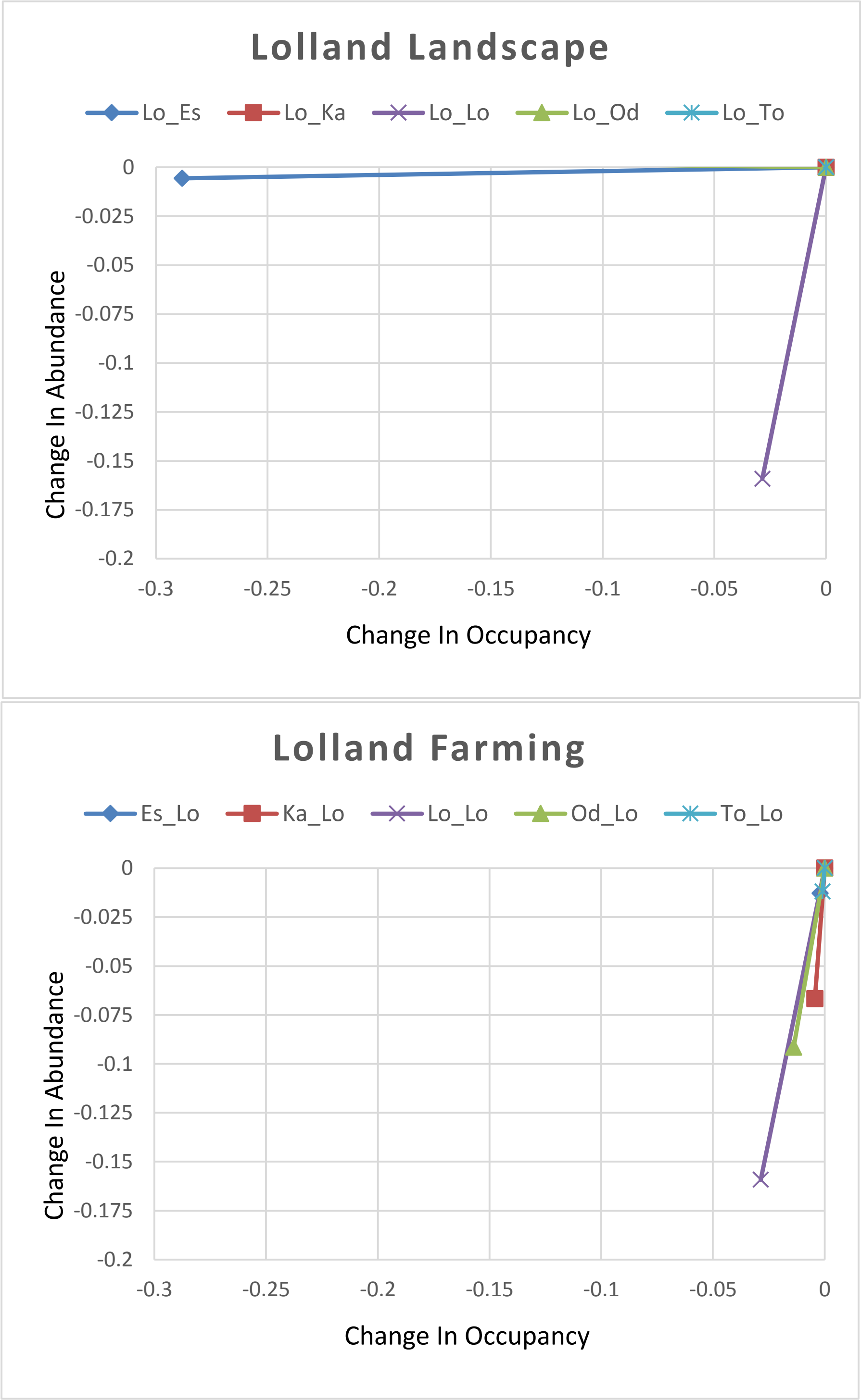


Figure 9: AOR-plots for hare ERA on Lolland landscape with Esbjerg, Karup, Lolland, Odder and Tønder farming (above) and Lolland farming on Esbjerg, Karup, Lolland, Odder and Tønder landscapes (below).

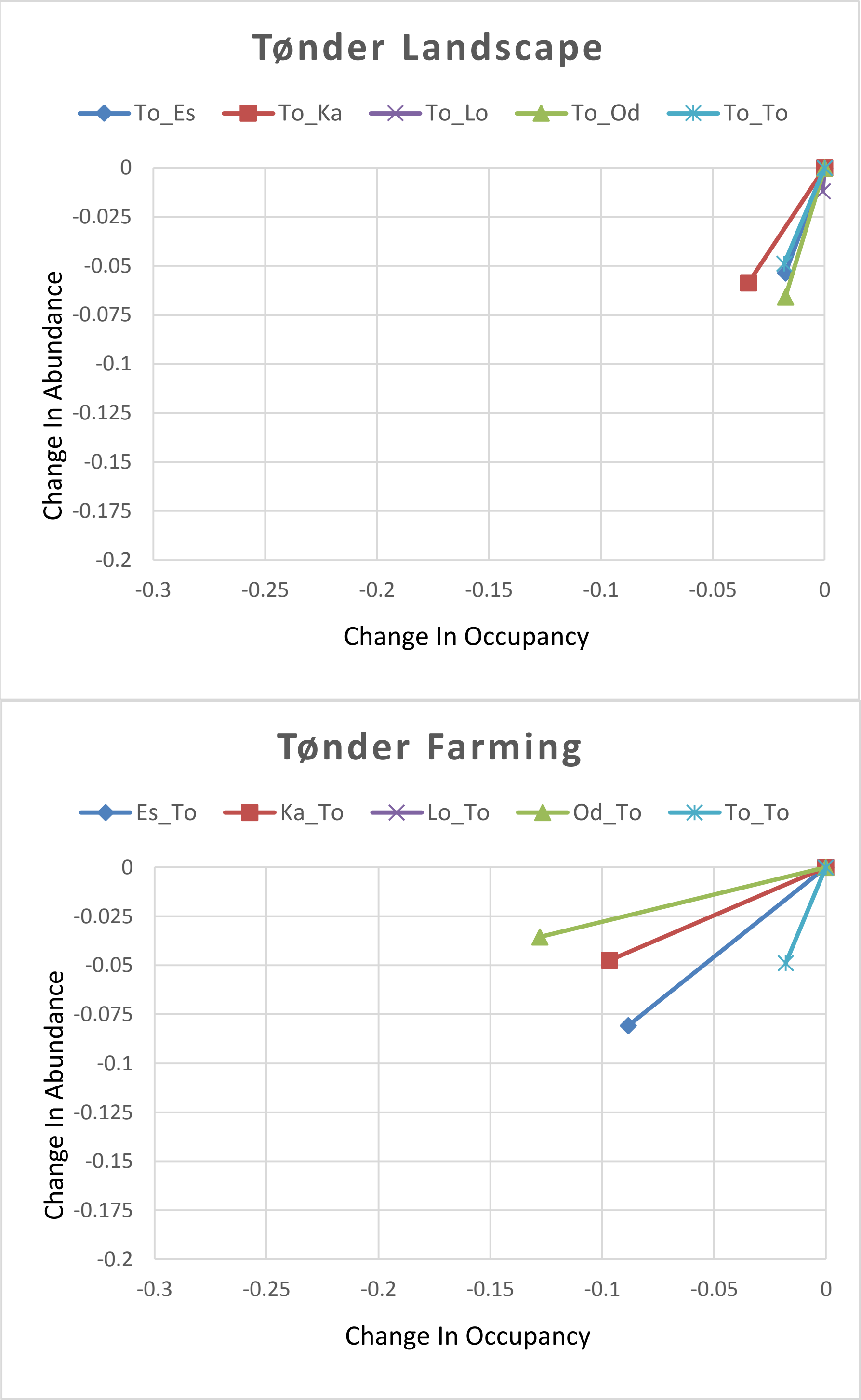


Figure 10: AOR-plots for hare ERA on Tønder landscape with Esbjerg, Karup, Lolland, Odder and Tønder farming (above) and Tønder farming on Esbjerg, Karup, Lolland, Odder and Tønder landscapes (below).

5.2 Bembidion lampros ERA

5.2.1 Standard scenarios

Beetle densities varied from 2.87 females m^{-2} in Lolland to 5.19 females m^{-2} in Tønder. Densities in the spring barley monoculture baseline were similar though not well correlated from 3.12 females m^{-2} in Toftlund to 4.29 in Tønder (Table 11).

Baseline	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Standard	4.13	4.52	3.62	3.58	2.87	3.59	3.36	3.58	3.49	5.19
SB100%	3.67	4.01	3.69	3.59	3.21	3.46	3.25	3.58	3.12	4.29

Table 11: The mean landscape density of female beetles m^{-2} in the last 10 years of simulation (years 21-20) predicted for the standard baseline and 100% spring barley baseline scenario for all landscapes (Es-Tn). This is the abundance score from the AOR-index.

Increasing toxicity from X1 to X1000 increased the impact of the pesticide, with X1000 having 3.9 to 5.0 times the impact of the X1 scenario (Table 12). Relative increase in impact was greatest for Lolland and Næstved and least for Tønder (Figure 11 & Table 13). Impact was correlated loosely with area treated for X10 toxicity scenarios (Figure 12), in most cases splitting the landscapes into two groups with high and lower impact. There was, however, also a significant negative relationship between population density expressed as Abundance from the AOR_Index and impact ($p = 0.015$, $df = 9$, $R^2 = 0.545$). However, there was one outlier; Toftlund landscape had a low population size, but also a low impact (Figure 13). There was no relationhip between total landscape abundance and relative effect ($R^2 = 0.0794$), indicating that the local field density of the beetle was more critical than the overall landscape density.

Monoculture spring barley scenarios had the greatest impact, and in SB100X10 all populations were driven to near extinction.

	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	0.914	0.908	0.880	0.847	0.778	0.830	0.793	0.844	0.923	0.961
WSOx10	0.760	0.781	0.698	0.608	0.442	0.558	0.497	0.592	0.743	0.893
WSOx100	0.716	0.740	0.578	0.497	0.259	0.415	0.354	0.470	0.710	0.839
WSOx1000	0.620	0.661	0.450	0.404	0.165	0.316	0.240	0.377	0.620	0.800
WSX10	0.750	0.790	0.713	0.615	0.441	0.587	0.523	0.603	0.761	0.891
Sx10	0.804	0.831	0.727	0.688	0.533	0.671	0.606	0.680	0.788	0.906
Wx10	0.823	0.830	0.769	0.639	0.480	0.595	0.531	0.635	0.803	0.899
Ox10	0.863	0.852	0.816	0.709	0.573	0.679	0.631	0.709	0.834	0.916
SBX10	0.331	0.363	0.371	0.295	0.258	0.346	0.299	0.291	0.245	0.199
SB100X10	0.006	0.030	0.014	0.012	0.005	0.010	0.003	0.006	0.002	0.056

Table 12: Mean female beetle population size relative to baseline for the last ten years of simulation (years 11-20 of pesticide application) for all standard scenarios and landscapes (Es-Tn).

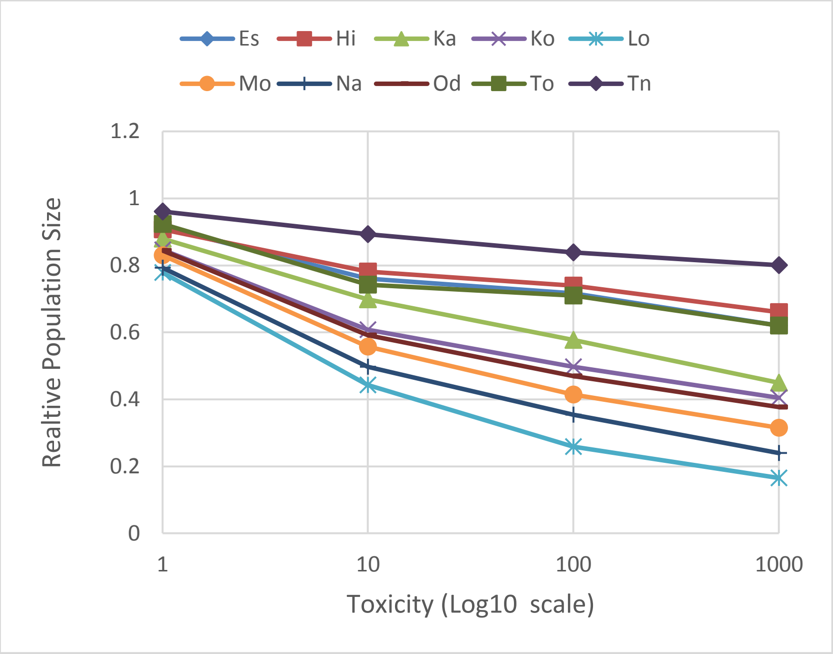


Figure 11: Changing impact of pesticides on beetle populations with increasing toxicity by factors of 10 for all ten standard landscapes assuming application to winter wheat, spring barley and winter oil seed rape.

	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Initial Effect	-0.086	-0.092	-0.120	-0.153	-0.222	-0.170	-0.207	-0.156	-0.077	-0.039
Slope	-0.093	-0.078	-0.141	-0.144	-0.202	-0.169	-0.180	-0.152	-0.094	-0.054

Table 13: The initial impact of X1 toxicity exposure and the rate of change in population depression per X10 increase in toxicity for all 10 landscapes assuming application to winter wheat, spring barley and oilseed rape for beetles. Slope fitted using linear estimation over 1X to1000X.

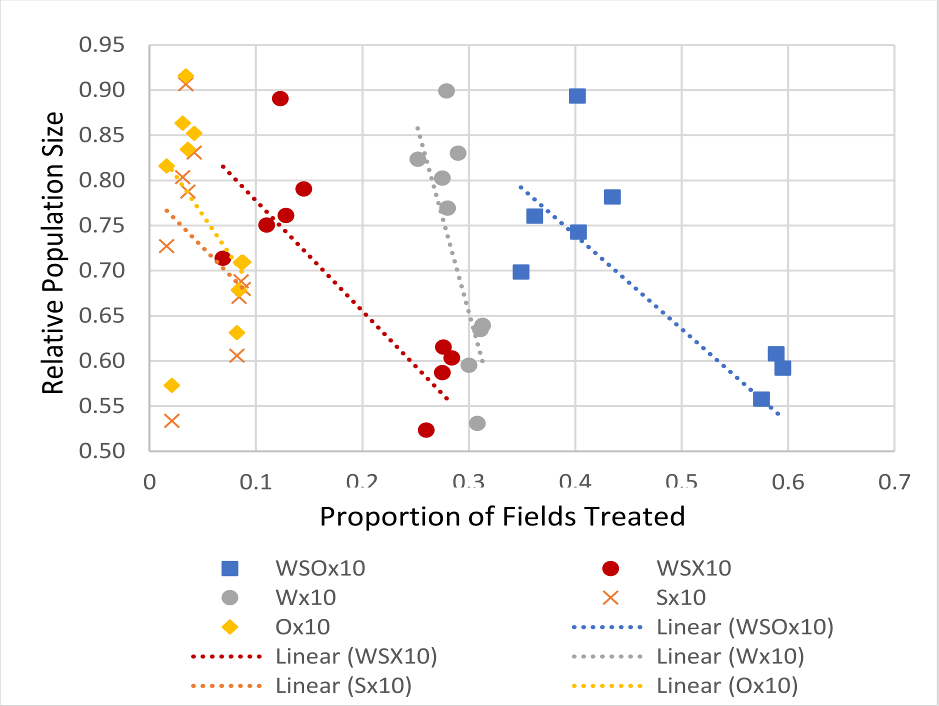


Figure 12: Relative Beetle population size plotted against area treated for WSOX10, WSX10, WX10, SX10 & OX10 scenarios with fitted regression lines.

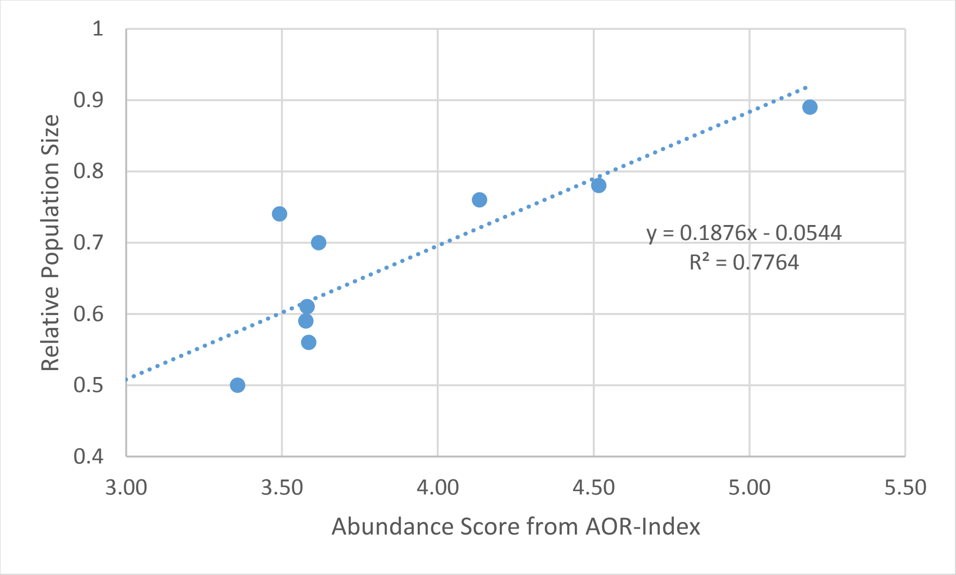


Figure 13: Impact plotted against beetle density (Abundance from AOR-Index) with a fitted regression line for the WSOX10 scenario.

Relative pgr suggested that in the majority of cases the beetle population decline had stabilised after 10 years of pesticide application (Table 14). However, in the case of SB100X10 the stable population was effectively zero. In Esbjerg, Kolding relative pgr showed no decline in the second decade of application. In Himmerland, Odder and Tønder declines in the second decade were only seen in the monoculture scenario. The other landscapes had declines in more, scenarios but with no clear pattern, but in some cases where no further decline was seen impacts were large e.g. Lolland WSOX1000.

Scenario	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1			-0.007			-0.007	-0.010			
WSOx10			-0.017			-0.013	-0.014		-0.014	
WSOx100			-0.019			-0.017				
WSOx1000			-0.016				-0.014			
WSX10			-0.012						-0.008	
Sx10			-0.019				-0.015		-0.013	
Wx10						-0.016	-0.023			
Ox10						-0.015	-0.018		-0.006	
SBX10										
SB100X10		-0.018						-0.018		-0.010

Table 14: For all standard scenarios and landscapes the fitted regression slopes for beetle population depressions relative to baseline for the second 10 years of application where the slope was significant at p<0.05.

Impacts on occupancy and abundance were relatively similar for all but the most extreme scenario when occupancy impacts increased relative to abundance (Table 15). However, some landscapes appeared to show a proportionally larger impact on occupancy (e.g. Ko, Lo, Na). In these cases the impacts were relatively higher suggesting that at higher impact there is a greater chance of local extinction.

	A/O	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	A	-0.06	-0.06	-0.08	-0.08	-0.11	-0.10	-0.12	-0.09	-0.04	-0.01
	O	-0.03	-0.03	-0.05	-0.08	-0.13	-0.08	-0.10	-0.08	-0.04	-0.03
WSOx10	A	-0.13	-0.11	-0.17	-0.16	-0.24	-0.22	-0.24	-0.18	-0.11	0.01
	O	-0.12	-0.12	-0.16	-0.28	-0.42	-0.29	-0.34	-0.28	-0.16	-0.12
WSOx100	A	-0.15	-0.12	-0.24	-0.19	-0.31	-0.29	-0.31	-0.23	-0.12	0.03
	O	-0.16	-0.16	-0.24	-0.39	-0.63	-0.42	-0.49	-0.39	-0.20	-0.19
WSOx1000	A	-0.22	-0.17	-0.31	-0.22	-0.32	-0.33	-0.37	-0.27	-0.16	0.05
	O	-0.21	-0.21	-0.35	-0.48	-0.76	-0.53	-0.62	-0.48	-0.26	-0.24
WSX10	A	-0.14	-0.10	-0.16	-0.15	-0.23	-0.20	-0.23	-0.18	-0.11	0.01
	O	-0.13	-0.12	-0.15	-0.27	-0.42	-0.27	-0.32	-0.27	-0.15	-0.12
Sx10	A	-0.11	-0.09	-0.16	-0.14	-0.21	-0.17	-0.20	-0.16	-0.10	0.00
	O	-0.09	-0.09	-0.13	-0.20	-0.32	-0.19	-0.24	-0.19	-0.12	-0.10
Wx10	A	-0.10	-0.09	-0.14	-0.16	-0.23	-0.21	-0.24	-0.17	-0.09	0.01
	O	-0.09	-0.09	-0.11	-0.24	-0.38	-0.25	-0.30	-0.23	-0.12	-0.11
Ox10	A	-0.08	-0.09	-0.11	-0.14	-0.20	-0.18	-0.20	-0.15	-0.08	0.00
	O	-0.06	-0.07	-0.08	-0.18	-0.28	-0.17	-0.21	-0.17	-0.09	-0.09
SBX10	A	-0.40	-0.38	-0.38	-0.41	-0.41	-0.36	-0.38	-0.42	-0.41	-0.39
	O	-0.45	-0.41	-0.40	-0.50	-0.56	-0.46	-0.51	-0.50	-0.58	-0.68
SB100X10	A	-0.48	-0.31	-0.48	-0.42	-0.32	-0.41	-0.33	-0.47	-0.38	0.27
	O	-0.99	-0.95	-0.97	-0.98	-0.99	-0.98	-1.00	-0.99	-1.00	-0.95

Table 15: AOR-index scores for beetles under all standard scenarios and landscapes (Es-Tn).

5.2.2 Life-stage scenarios

Impacts on population size of the LX10 scenario varied from 3.4% to 21.5%, and the adult only effects scenario from 4.1% to 23.3%. Occupancy for the LX10 and A scenarios ranged from 3.5% to 15.2% with a similar range but differing distribution between the two scenarios. Impacts of the combined ALX10 scenario varied between landscapes from 7% to 42% reduction in population size, 7% to 31% reduction in occupancy and 0% to 16% reduction in abundance. For all three scenarios, and as with other species and scenarios the impact varied depending upon the landscape, some tending to large decreases in occupancy others decreasing more in abundance (Table 16).

In all cases, impacts of the ALX10 scenario were lower than summed effect of the two scenarios run separately. The difference was between 2.5% and 22.4% of the ALX10 effect depending on the landscape (Figure 14). The difference was, however, not a constant proportion compared amongst landscapes; for example, Lolland and Tønder had similar magnitude of differences but very different relative differences. This indicates that landscape has a large effect which also interacts with the differences related to running these scenarios as a single combined scenario or separately.

Landscape	Population Size			Occupancy			Abundance		
	A	**LX10**	**ALX10**	**A**	**LX10**	**ALX10**	**A**	**LX10**	**ALX10**
Es	-0.107	-0.049	-0.141	-0.041	-0.017	-0.056	-0.068	-0.033	-0.089
Hi	-0.090	-0.034	-0.120	-0.038	-0.013	-0.052	-0.054	-0.021	-0.071
Ka	-0.117	-0.081	-0.189	-0.048	-0.034	-0.087	-0.071	-0.049	-0.111
Ko	-0.162	-0.103	-0.249	-0.091	-0.059	-0.156	-0.077	-0.047	-0.108
Lo	-0.233	-0.215	-0.420	-0.152	-0.146	-0.308	-0.094	-0.082	-0.159
Mo	-0.187	-0.102	-0.272	-0.092	-0.049	-0.149	-0.103	-0.056	-0.143
Na	-0.195	-0.139	-0.323	-0.102	-0.072	-0.189	-0.102	-0.074	-0.163
Od	-0.164	-0.096	-0.251	-0.088	-0.053	-0.150	-0.082	-0.045	-0.117
To	-0.105	-0.068	-0.141	-0.059	-0.037	-0.083	-0.049	-0.033	-0.062
Tn	-0.041	-0.035	-0.073	-0.035	-0.034	-0.074	-0.006	-0.001	0.001

Table 16: The relative impact in terms of population size, occupancy and abundance when comparing three beetle toxicity scenarios (A = adult toxicity, LX10 = larval toxicity at 10X sensitivity than adult, and ALX10 combines both in the same simulation).

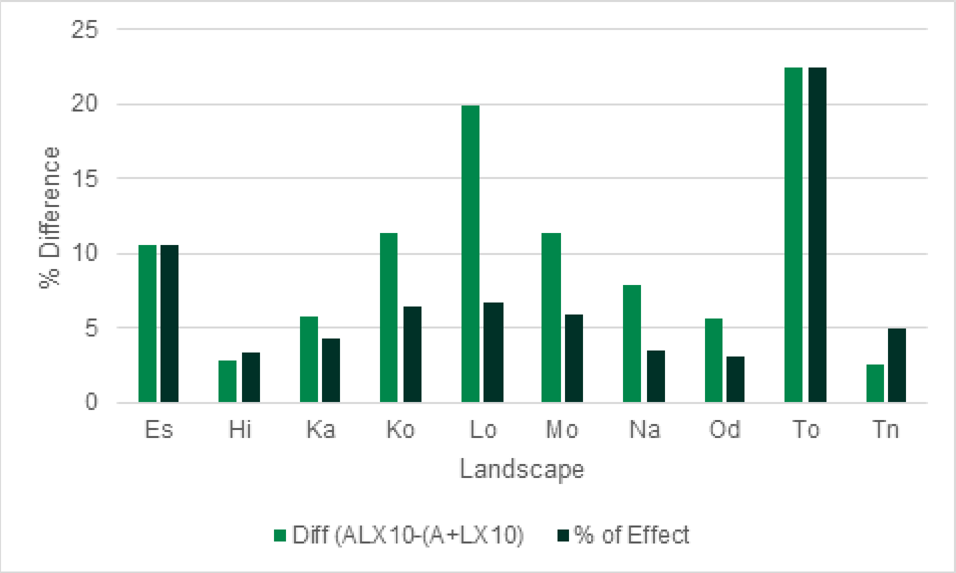


Figure 14: Difference between ALX10 effects and the sum of LX10 + A scenarios run separately as a percentage of the effect, and the same value scaled to the maximum percentage for comparison of pattern.

5.3 Skylark ERA

5.3.1 Standard scenarios

The baseline densities for skylarks varied considerably between landscapes, highest densities in Lolland and lowest in Karup. Densities in the spring barley monoculture baselines was higher than the standard baseline in all cases except Lolland (Table 17).

Impacts of the standard scenarios were relatively low compared to other species. The largest impact was seen from the WSOX1000 scenario in the Esbjerg landscape (-31%) (Table 18). In Tønder landscape impacts were extremely low, and only seen at all in monoculture spring barley scenarios.

Baseline	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Standard	21.11	24.62	12.41	24.40	44.19	25.53	28.10	27.98	30.78	34.82
SB100%	23.23	26.58	12.67	28.03	39.21	29.14	32.49	32.54	34.35	40.59

Table 17: The mean landscape density of female skylarks km^{-2} in the last 10 years of simulation (years 21-20) predicted for the standard baseline and 100% spring barley baseline scenario for all landscapes (Es-Tn).

	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	0.82	0.91	0.85	0.95	0.97	0.92	0.97	0.97	0.91	1.00
WSOx10	0.75	0.88	0.79	0.93	0.95	0.90	0.95	0.96	0.87	1.00
WSOx100	0.72	0.85	0.75	0.92	0.94	0.88	0.95	0.95	0.84	1.00
WSOx1000	0.69	0.83	0.73	0.92	0.94	0.87	0.94	0.93	0.81	1.00
WSX10	0.78	0.89	0.79	0.94	0.96	0.92	0.96	0.96	0.89	1.00
Sx10	0.88	0.96	0.84	0.99	1.00	0.98	1.00	0.99	0.95	1.00
Wx10	0.84	0.94	0.88	0.96	0.97	0.94	0.98	0.97	0.92	1.00
Ox10	0.95	0.99	0.93	1.00	1.00	1.00	1.00	1.00	0.98	1.00
SBX10	0.78	0.88	0.84	0.94	0.97	0.94	0.97	0.96	0.89	0.97
SB100X10	0.80	0.91	0.83	0.95	1.00	0.96	0.99	0.98	0.92	0.99

Table 18: Mean female skylark population size relative to baseline for the last ten years of simulation (years 11-20 of pesticide application) for all standard scenarios and landscapes (Es-Tn).

Increasing toxicity did not increase the impact very greatly (Figure 15). Largest impacts were on Karup and Esbjerg landscape where impacts were doubled to 25% & 30% respectively. Slopes of increased response with increased toxicity were therefore very shallow (Table 19).

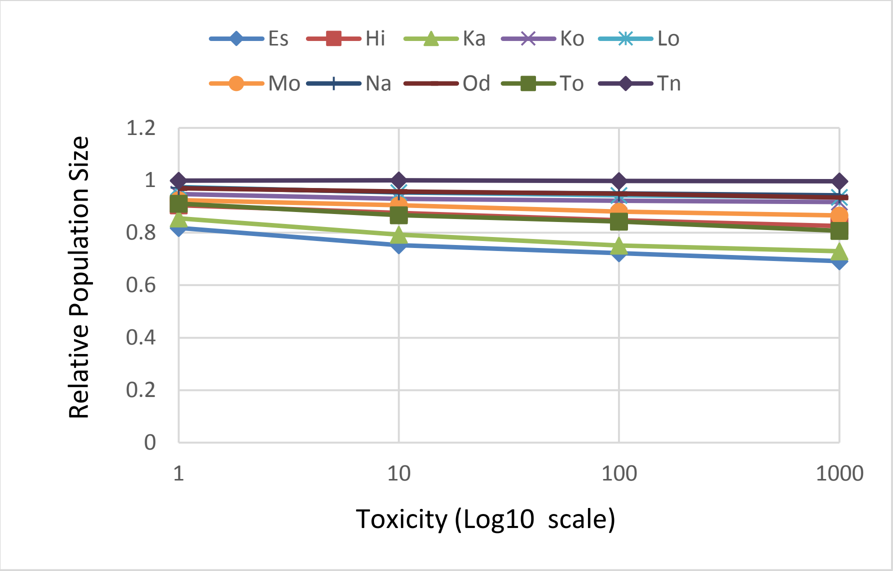


Figure 15: Changing impact of pesticides on skylarks with increasing toxicity by factors of 10 for all ten standard landscapes assuming application to winter wheat, spring barley and winter oil seed rape.

	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Initial Effect	-0.182	-0.095	-0.145	-0.052	-0.025	-0.076	-0.028	-0.030	-0.089	-0.001
Slope	-0.041	-0.026	-0.042	-0.010	-0.013	-0.020	-0.009	-0.011	-0.033	-0.001

Table 19: The initial impact of X1 toxicity exposure and the rate of change in skylark population depression per X10 increase in toxicity for all 10 landscapes assuming application to winter wheat, spring barley and oilseed rape. Slope fitted using linear estimation over 1X to1000X.

Unlike the beetle and hare, impacts on skylark populations did not increase with increasing area treated, in fact the opposite seemed the case for all X10 scenarios (Figure 16), although not statistically significant. There was, however, a negative relationship between impact and population size ($p = 0.025$, $df = 9$, $R^2 = 0.4844$) (Figure 17).

Relative population growth rate showed population declines in only three landscapes, and of those Himmerland only had shallow decline in the WSOX100 scenario. Esbjerg & Karup exhibited declines in most scenarios (Table 20).

Where impacts were evident, these were primarily reductions in occupancy rather than abundance (Table 21).

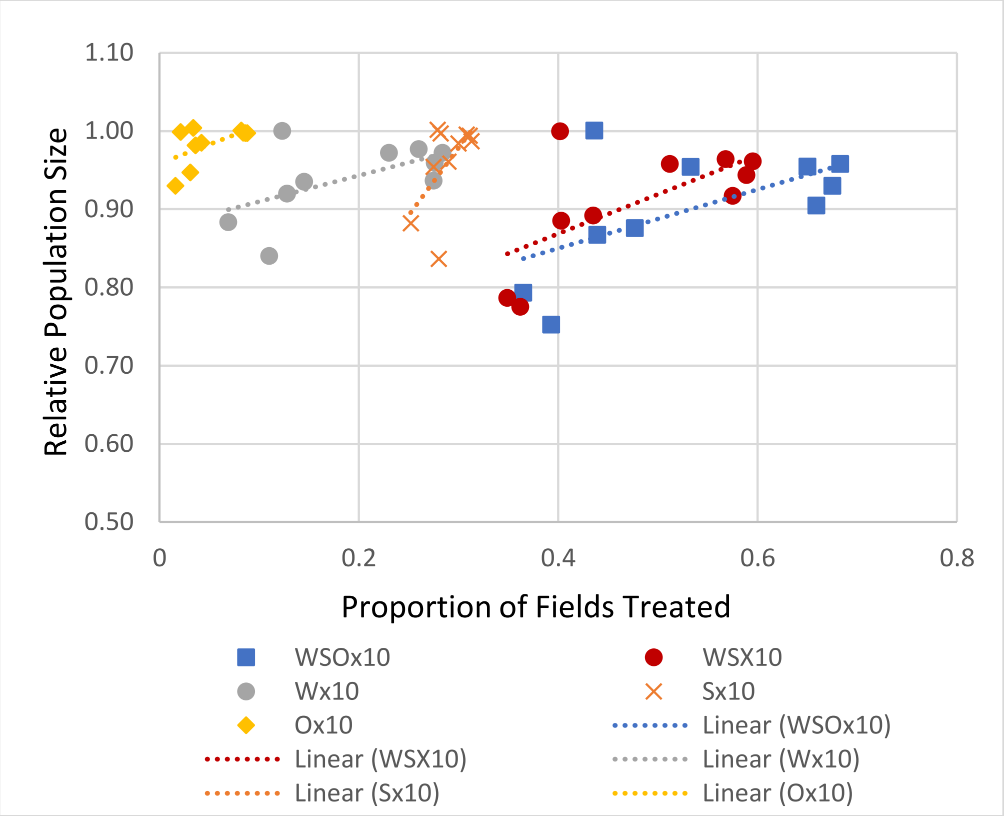


Figure 16: Relative skylark population size plotted against area treated for all landscapes for scenarios WSOX10, WSX10, WX10, SX10 & OX10, with fitted regression line.

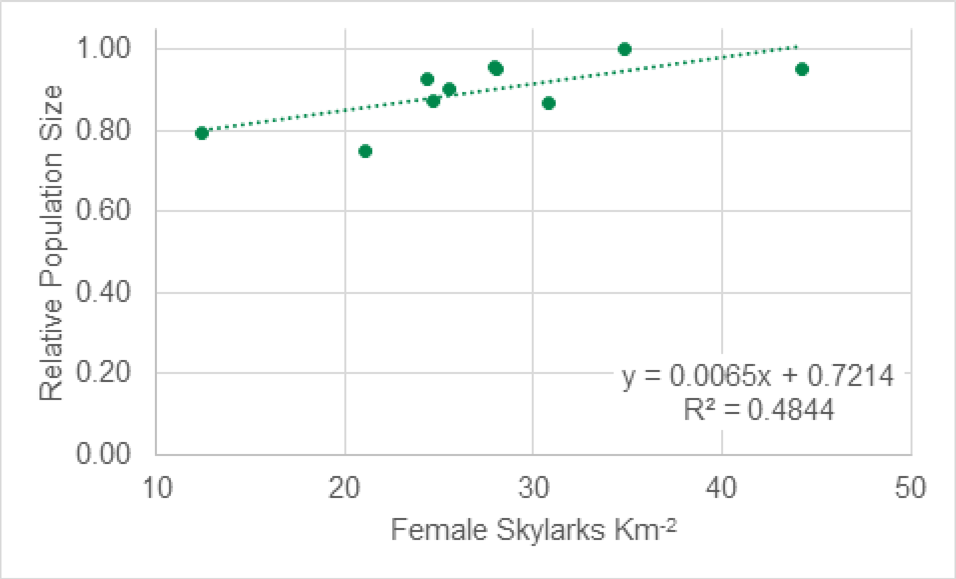


Figure 17: Relative skylark population size for WSOX10 scenario plotted against baseline population size, with fitted regression line.

Scenario	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	-0.006		-0.006							
WSOx10	-0.010		-0.006							
WSOx100	-0.012		-0.007							
WSOx1000	-0.012	-0.008	-0.008							
WSX10	-0.007		-0.006							
Sx10			-0.005							
Wx10	-0.005									
Ox10										
SBX10										
SB100X10										

Table 20: The fitted regression slopes for skylark population depressions relative to baseline for the second 10 years of application where the slope was significant at $p<0.05$ for all standard scenarios and landscapes.

	A/O	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	A	-0.05	-0.04	-0.03	-0.03	-0.03	-0.04	-0.02	-0.02	-0.04	0.00
	O	-0.14	-0.06	-0.12	-0.02	0.00	-0.04	-0.01	-0.01	-0.05	0.00
WSOx10	A	-0.07	-0.05	-0.04	-0.04	-0.05	-0.05	-0.03	-0.03	-0.06	0.00
	O	-0.19	-0.08	-0.17	-0.03	0.00	-0.05	-0.01	-0.01	-0.08	0.00
WSOx100	A	-0.07	-0.05	-0.04	-0.04	-0.06	-0.06	-0.03	-0.04	-0.07	0.00
	O	-0.22	-0.10	-0.22	-0.04	0.00	-0.07	-0.02	-0.02	-0.09	0.00
WSOx1000	A	-0.08	-0.06	-0.04	-0.04	-0.06	-0.06	-0.04	-0.04	-0.09	0.00
	O	-0.25	-0.12	-0.24	-0.04	0.00	-0.08	-0.02	-0.03	-0.12	0.00
WSX10	A	-0.06	-0.04	-0.04	-0.03	-0.04	-0.04	-0.03	-0.03	-0.06	0.00
	O	-0.18	-0.07	-0.18	-0.03	0.00	-0.04	-0.01	-0.01	-0.06	0.00
Sx10	A	-0.03	-0.02	-0.03	-0.01	0.00	-0.01	0.00	0.00	-0.02	0.00
	O	-0.09	-0.02	-0.14	-0.01	0.00	-0.01	0.00	0.00	-0.03	0.00
Wx10	A	-0.04	-0.03	-0.02	-0.02	-0.03	-0.03	-0.02	-0.02	-0.04	0.00
	O	-0.12	-0.04	-0.10	-0.02	0.00	-0.03	-0.01	-0.01	-0.04	0.00
Ox10	A	-0.01	-0.01	-0.01	0.00	0.00	0.00	0.00	0.00	-0.01	0.00
	O	-0.04	-0.01	-0.06	0.00	0.00	0.00	0.00	0.00	-0.01	0.00
SBX10	A	-0.09	-0.06	-0.05	-0.04	-0.03	-0.04	-0.03	-0.04	-0.07	-0.02
	O	-0.14	-0.06	-0.12	-0.02	0.01	-0.02	0.00	0.00	-0.04	0.00
SB100X10	A	-0.09	-0.06	-0.06	-0.05	-0.03	-0.04	-0.03	-0.04	-0.06	-0.01
	O	-0.11	-0.03	-0.12	0.00	0.03	0.00	0.02	0.02	-0.02	0.00

Table 21: Relative AOR-index scores for skylarks under all standard scenarios and landscapes (Es-Tn). A = Abundance O = Occupancy

5.3.2 Eggshell thickness reduction scenarios

The worst possible theoretical scenario would be if all skylarks failed to breed due to eggshell cracking. In order to better interpret the later scenarios the impact of this scenario was also simulated (Figure 18). Here extinction of the population occurs after 5 years, which represents the length of time the previous generation can maximally live for. Any successful breeding will extend the length of time taken to reach this point. This scenario therefore acts as a back-stop for effect, impacts on reproduction cannot be worse than this.

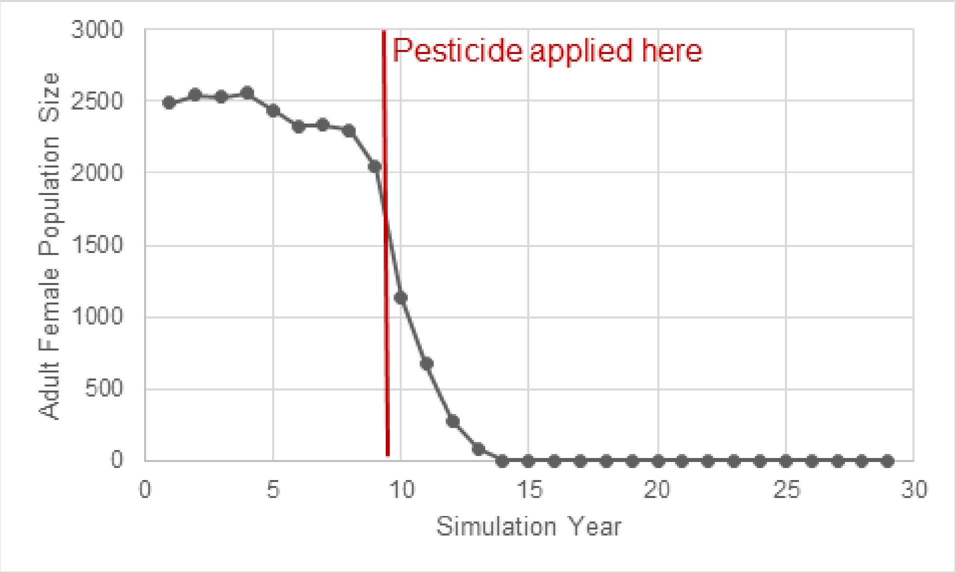


Figure 18: Changes in total adult female population numbers assuming all skylarks were prevented from breeding due to clutch failure after the application of pesticide starting in the breeding season of year 10. Population size measured at the end of the each year.

5.3.2.1 Eggshell Scenario 1

The purpose of this scenario was to find out which (if any) landscape had the most sensitive skylark population. Assuming global exposure (i.e. all birds were assumed to forage from pesticide treated fields), and assuming that egg shell reduction results in clutch loss at 5% - 75% chance, the impact relative to the baseline population size was determined for all 10 landscapes. At 5%, 5 out of 10 landscapes showed no population impact, and a maximum impact of 2% in the other 5. Impact increased with percentage clutch loss chance and at 50% virtual extinction in Esbjerg, Himmerland, Mors and Toftland was observed, at 75% all populations were extinct before the end of 20 years of application.

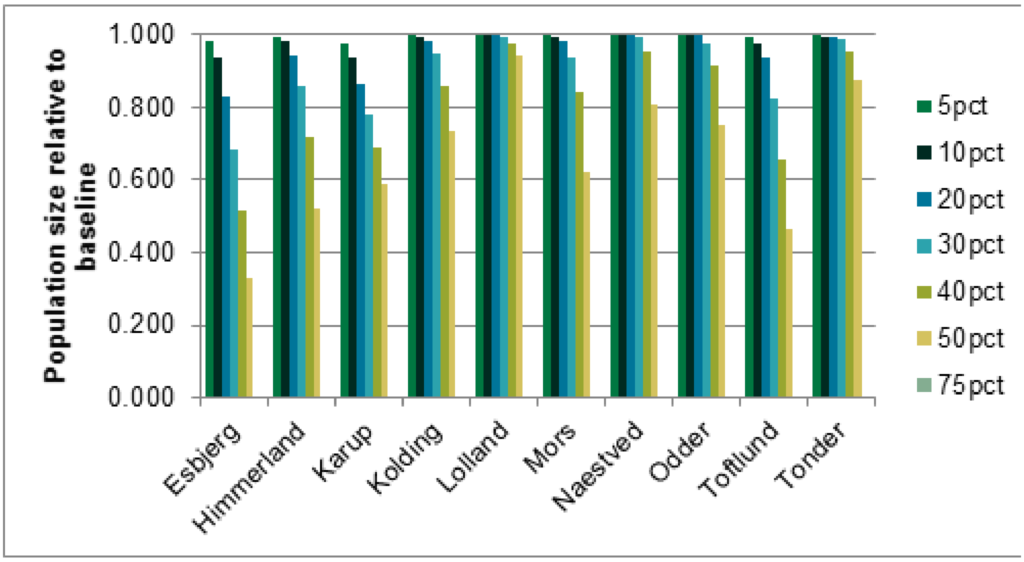


Figure 19: Skylark eggshell thickness reduction effects in terms of population size over years 20-30 of simulation relative to baseline for all ten landscapes. Scenario assumptions are that all skylarks were affected by a fixed percentage chance of total clutch loss (5-75pct). Note that for 75pct all populations went extinct before the end of 20 years of application.

The mean change in population relative to baseline was calculated for each landscape across all 7-percentage effects and plotted against population size (Figure 20). There was a clear relationship between population size and effect, with higher population sizes tending to lower relative population impacts, but the population size is clearly only one determinant of the impact. The Esbjerg landscape appeared to have the most sensitive populations of skylarks, hence Esbjerg was chosen for use in Eggshell Scenarios 2-4.

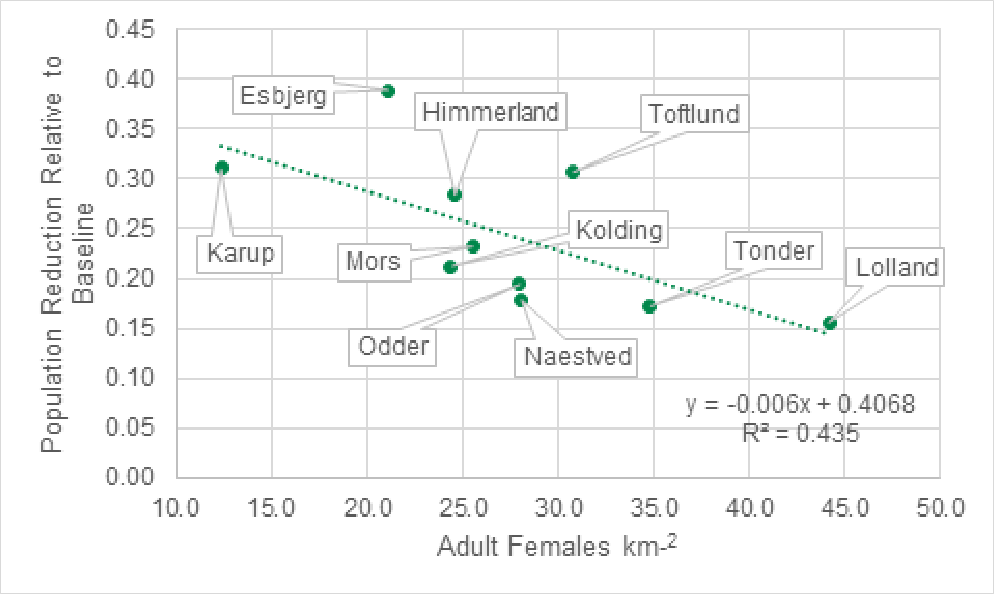


Figure 20: Size of the mean population reduction relative to the baseline per landscape taken as a mean effect across all effect levels in the Eggshell Scenario 1 group with a fitted linear regression line.

When comparing final year AOR-index effects in Esbjerg, extinction occurred in all replicates of global exposure and 75% chance of clutch loss and only 5% and 10% chance scenarios produced stable population after 20 years of application (Table 22), with continuing declines in populations above 10% clutch loss (Table 23). Long-term extinction was therefore the predicted result in 20% and upwards chance of clutch loss.

	Effect Level						
AOR-index component	5%	10%	20%	30%	40%	50%	75%
Abundance	-0.006	-0.014	-0.037	-0.070	-0.111	-0.169	-1.000
Occupancy	-0.014	-0.051	-0.137	-0.266	-0.417	-0.600	-1.000

Table 22: AOR index scores for skylarks in Esbjerg landscape assuming global eggshell thinning effects affecting whole clutches with 5-75% probability.

Effect Level	Years 11-20	Years 21-20
5pct	-0.001	0.000
10pct	-0.008	0.000
20pct	-0.017	-0.007
30pct	-0.028	-0.012
40pct	-0.042	-0.016
50pct	-0.056	-0.017
75pct	-0.057	Extinct

Table 23: Slopes of the population trajectory relative to the baseline for the 1st and 2nd decade of pesticide application assuming 5-75% clutch mortality on the Esbjerg landscape and global exposure.

5.3.2.2 Eggshell Scenario 2 & 3

Contrary to Eggshell Scenario 1, the impacts of the standard WSO scenario even with 100% application assumed (i.e. all crops apply three times) did not have a large impact on the skylark population. There was very little difference between assumptions of individual egg effects or fully clutch effects. Assuming a monoculture of spring barely decreased the impact compared to WSOX75X1 scenario, and in this case considerably so for egg effects (Table 24). These results indicate the potential for compensation to some degree in the model, since all things being otherwise equal the number of eggs affected by x% clutch effects and x% egg effects should be the same, only the distribution in space differs.

	AOR-index component	WSO50X1	WSO75X1	SB75X1	SB75X1000
Clutch Effects	Abundance	-0.06	-0.09	-0.01	-0.01
	Occupancy	-0.05	-0.11	-0.05	-0.05
Egg Effects	Abundance	-0.07	-0.07	-0.01	0.00
	Occupancy	-0.05	-0.07	-0.01	-0.01

Table 24: Impacts on abundance and occupancy of the skylark population in the Esbjerg landscape when assumed 50% (50X1) or 75% (75X1) clutch or egg loss when exposed to a threshold concentration of pesticide assuming 100% application to all winter wheat, spring barley and oilseed rape at X1 toxicity and otherwise following the standard scenario. SB75X1 & SB78X1000 assume a monoculture of spring barley with 3 applications of X1 or X1000 toxicity following the standard scenario but 100% probability of application.

5.3.2.3 Eggshell Scenario 4

Advancing the timing of spraying in spring barley by 4 weeks had a dramatic effect (Table 25). The effect can be compared between scenarios SB75X1 (Table 24) and 75X1A (Table 25), which are identical except for the spray timing. In all cases the impacts were calculated against the relevant baselines so can be compared directly. Advancing spraying increased impacts from -0.01 to -0.31 for clutch effects on abundance, and -0.01 to -0.26 for egg effects. Occupancy was similarly affected, -0.05 to 0.74 for clutch effects and -0.01 to -0.60 for egg effects. These effects are close to the impacts seen by global exposure for 75pct effect, but no extinction occurred. In this case, egg effects appear to have a bigger impact than whole clutch effects. This may indicate that there is some compensation for clutch effects operating in the model. This should be the case if whole clutches are lost early and can be re-laid. This second clutch would be subject to the same mortality percentage but the result is independent of the previous clutch mortality, hence some compensation is possible.

		10X1A	50X1A	75X1A	10X1000A	50X1000A	75X1000A
Clutch Effects	**A**	0.006	-0.068	-0.314	0.005	-0.074	-0.320
	O	-0.043	-0.173	-0.740	-0.040	-0.188	-0.759
Egg Effects	**A**	0.002	-0.166	-0.264	0.002	-0.186	-0.282
	O	-0.055	-0.350	-0.596	-0.052	-0.385	-0.631

Table 25: Skylark populations in the Esbjerg landscape assuming a monoculture of spring barley. Egg-shell thinning effects at 10, 50 & 75% and X1 & X1000 toxicity. Here spray applications are advanced by 4 weeks compared to the standard scenario to April and all sprays have a 100% chance of application. (A = abundance, O = occupancy).

5.4 Great Crested Newt ERA

5.4.1 Standard Scenarios

Baseline densities based on global optimal pond quality were very variable amongst the landscapes (Table 26). Lowest population size was in Karup and highest in Næstved, followed by Tønder. There was no difference in densities between the standard baseline and the spring barely monoculture baselines.

	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Baseline	69	58	47	91	83	40	139	72	77	122
SB100 Baseline	68	59	48	91	82	40	137	72	76	123

Table 26: The mean landscape density of female newt km^{-2} in the last 10 years of simulation (years 21-20) predicted for the standard baseline and 100% spring barley baseline scenario for all landscapes (Es-Tn).

Impacts of standard scenarios varied greatly (Table 27). Spring barley-monoculture scenarios having the greatest impact, with virtual extinction in Toftlund landscape, and very high impacts in Lolland, Næstved and Tønder. The population in Karup were the least affected in all scenarios.

	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	0.88	0.88	0.94	0.69	0.62	0.76	0.61	0.78	0.82	0.76
WSOx10	0.76	0.79	0.87	0.47	0.36	0.56	0.35	0.60	0.66	0.71
WSOx100	0.68	0.73	0.81	0.38	0.25	0.47	0.24	0.50	0.58	0.68
WSOx1000	0.62	0.68	0.76	0.33	0.20	0.41	0.18	0.45	0.52	0.66
WSX10	0.77	0.80	0.86	0.49	0.37	0.58	0.37	0.62	0.67	0.71
Sx10	0.83	0.85	0.89	0.65	0.57	0.72	0.56	0.75	0.77	0.74
Wx10	0.82	0.84	0.91	0.55	0.45	0.64	0.45	0.67	0.74	0.74
Ox10	0.88	0.89	0.93	0.73	0.68	0.78	0.65	0.81	0.83	0.76
SBX10	0.41	0.41	0.60	0.25	0.16	0.33	0.17	0.36	0.15	0.11
SB100X10	0.24	0.28	0.45	0.14	0.08	0.20	0.08	0.22	0.04	0.06

Table 27: Mean female newt population size relative to baseline for the last ten years of simulation (i.e. years 11-20 of pesticide application) for all standard scenarios and landscapes (Es-Tn).

Slopes of increased impact with toxicity were relatively shallow (Figure 21) with a maximum change in impact of four times in Karup, and minimum of 1.4 times in Tønder. However, compared to other species the initial impact of the X1 scenario was much higher, up to 39% in Næstved (Table 28).

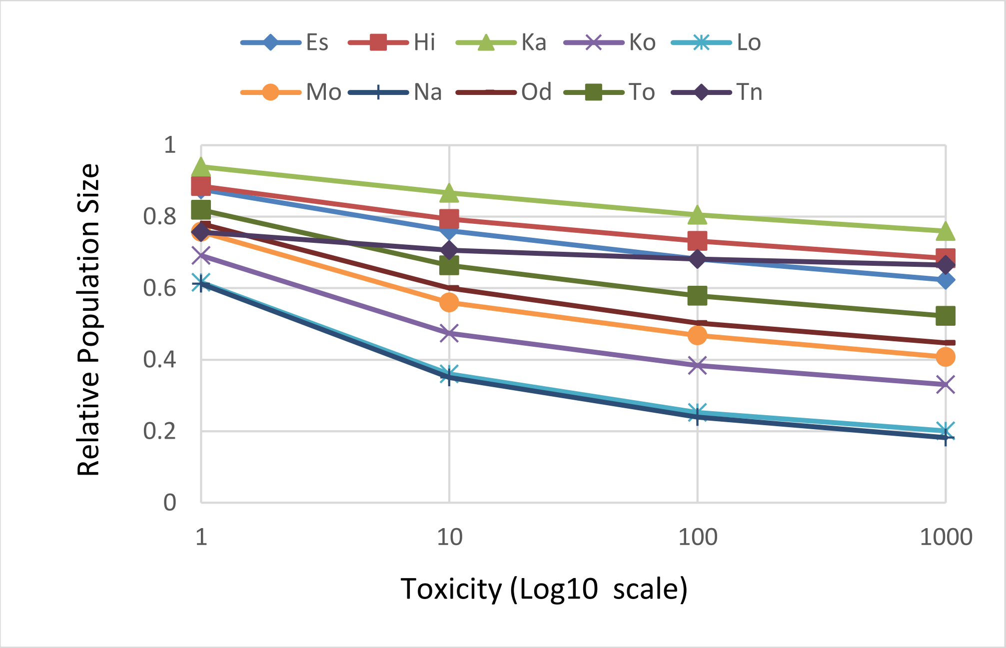


Figure 21: Changing impact of pesticides on newt populations with increasing toxicity by factors of 10 for all ten standard landscapes assuming application to winter wheat, spring barley and winter oil seed rape.

	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Initial Effect	-0.123	-0.115	-0.060	-0.309	-0.384	-0.242	-0.388	-0.220	-0.181	-0.243
Slope	-0.084	-0.067	-0.060	-0.117	-0.135	-0.114	-0.140	-0.110	-0.098	-0.030

Table 28: The initial impact of X1 toxicity exposure and the rate of change in newt population depression per X10 increase in toxicity for all 10 landscapes assuming application to winter wheat, spring barley and oilseed rape. Slope fitted using linear estimation over 1X to 1000X.

Area treated was a major determinant of the impact at population level with very steep slopes, although there was considerable scatter (Figure 22). Across all X10 applications within a landscape the impact was strongly related to the area (e.g. $y = -3.04x + 2.69$, $R^2 = 0.842$ for Esbjerg). The relationship between total population size and impact was positive (Figure 23). There was no correlation between baseline abundance score and impact amongst landscapes (e.g. WSOX10 $R^2 = 0.001$).

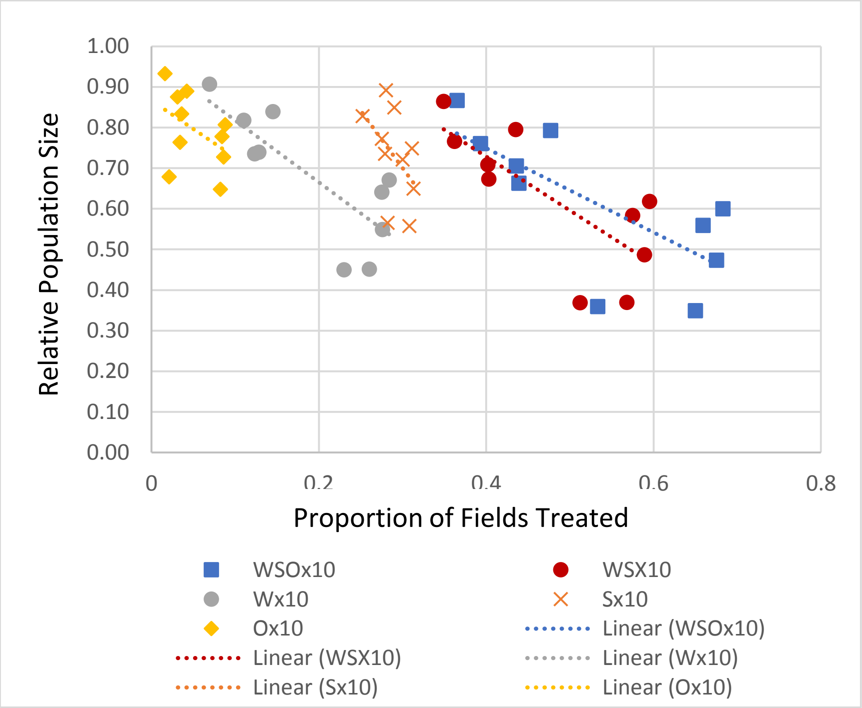


Figure 22: Relative newt population size plotted against area treated for WSOX10, WSX10, WX10, SX10 & OX10 scenarios with fitted regression lines.

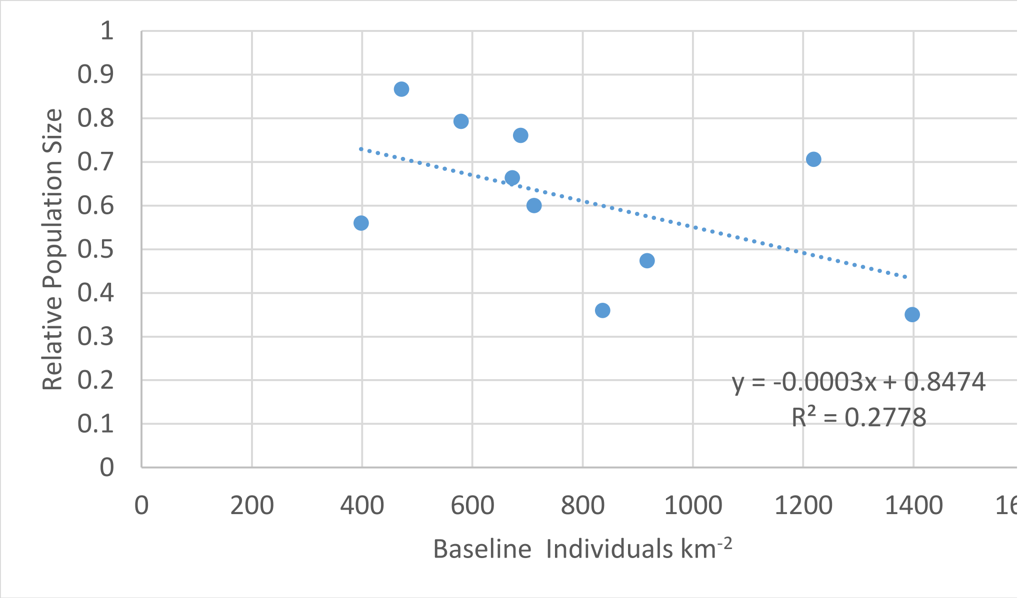


Figure 23: Relative newt population size for WSOX10 scenario plotted against baseline population size, with fitted regression line.

The newt model was unique in that in all cases the population in the second decade of application showed no declines relative to the baseline population, i.e. all landscape-scale populations were stable. AOR-index scores for the scenarios indicate that the newt population survived the X1 application scenarios although being reduced in terms of abundance, and in some cases dramatically (e.g. Næstved by 37%), but occupancy was relatively stable. Monoculture scenarios however resulted in very large impacts at X10 toxicity, including 35-72% decrease in occupancy. The impact of the SB100X10 scenario was approximately to double the reduction in occupancy for all landscapes.

	A/O	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
WSOx1	A	-0.09	-0.08	-0.04	-0.28	-0.35	-0.18	-0.37	-0.18	-0.16	-0.32
	O	-0.03	-0.04	-0.02	-0.05	-0.05	-0.07	-0.02	-0.05	-0.03	0.11
WSOx10	A	-0.18	-0.14	-0.09	-0.47	-0.59	-0.33	-0.62	-0.33	-0.29	-0.35
	O	-0.07	-0.07	-0.05	-0.11	-0.12	-0.17	-0.07	-0.11	-0.07	0.09
WSOx100	A	-0.24	-0.18	-0.13	-0.54	-0.69	-0.38	-0.73	-0.40	-0.36	-0.36
	O	-0.10	-0.11	-0.07	-0.17	-0.19	-0.24	-0.13	-0.16	-0.10	0.07
WSOx1000	A	-0.28	-0.22	-0.17	-0.58	-0.73	-0.42	-0.78	-0.44	-0.41	-0.37
	O	-0.13	-0.13	-0.09	-0.22	-0.25	-0.30	-0.19	-0.20	-0.12	0.06
WSX10	A	-0.17	-0.14	-0.09	-0.46	-0.58	-0.31	-0.60	-0.31	-0.28	-0.35
	O	-0.07	-0.08	-0.05	-0.10	-0.12	-0.16	-0.06	-0.10	-0.07	0.09
Sx10	A	-0.13	-0.10	-0.07	-0.31	-0.40	-0.21	-0.43	-0.21	-0.20	-0.33
	O	-0.05	-0.05	-0.04	-0.05	-0.06	-0.08	-0.03	-0.05	-0.04	0.10
Wx10	A	-0.14	-0.11	-0.06	-0.40	-0.51	-0.27	-0.53	-0.27	-0.23	-0.33
	O	-0.05	-0.05	-0.03	-0.08	-0.08	-0.12	-0.04	-0.08	-0.04	0.10
Ox10	A	-0.10	-0.08	-0.05	-0.25	-0.29	-0.17	-0.34	-0.16	-0.15	-0.31
	O	-0.03	-0.03	-0.02	-0.03	-0.04	-0.06	-0.02	-0.04	-0.02	0.11
SBX10	A	-0.42	-0.41	-0.28	-0.67	-0.77	-0.50	-0.79	-0.52	-0.77	-0.87
	O	-0.29	-0.29	-0.17	-0.25	-0.30	-0.34	-0.21	-0.24	-0.33	-0.18
SB100X10	A	-0.43	-0.40	-0.31	-0.67	-0.72	-0.46	-0.79	-0.55	-0.84	-0.81
	O	-0.58	-0.53	-0.35	-0.56	-0.70	-0.63	-0.64	-0.51	-0.72	-0.70

Table 29: AOR-index scores for newts under all standard scenarios and landscapes (Es-Tn).

5.4.2 Overspray Scenarios

Not surprisingly, the newt overspray scenarios produced a lower impact on the newt populations compared to the long-term environmental exposure in the standard scenarios. Time series of the impact show the importance of the weather in terms of deviation from baseline conditions, scaling with increasing LCx (Figure 24). The same figure also demonstrates an important point; the LC5 scenario appears to have no effect on the population until the after the 3rd year of application.

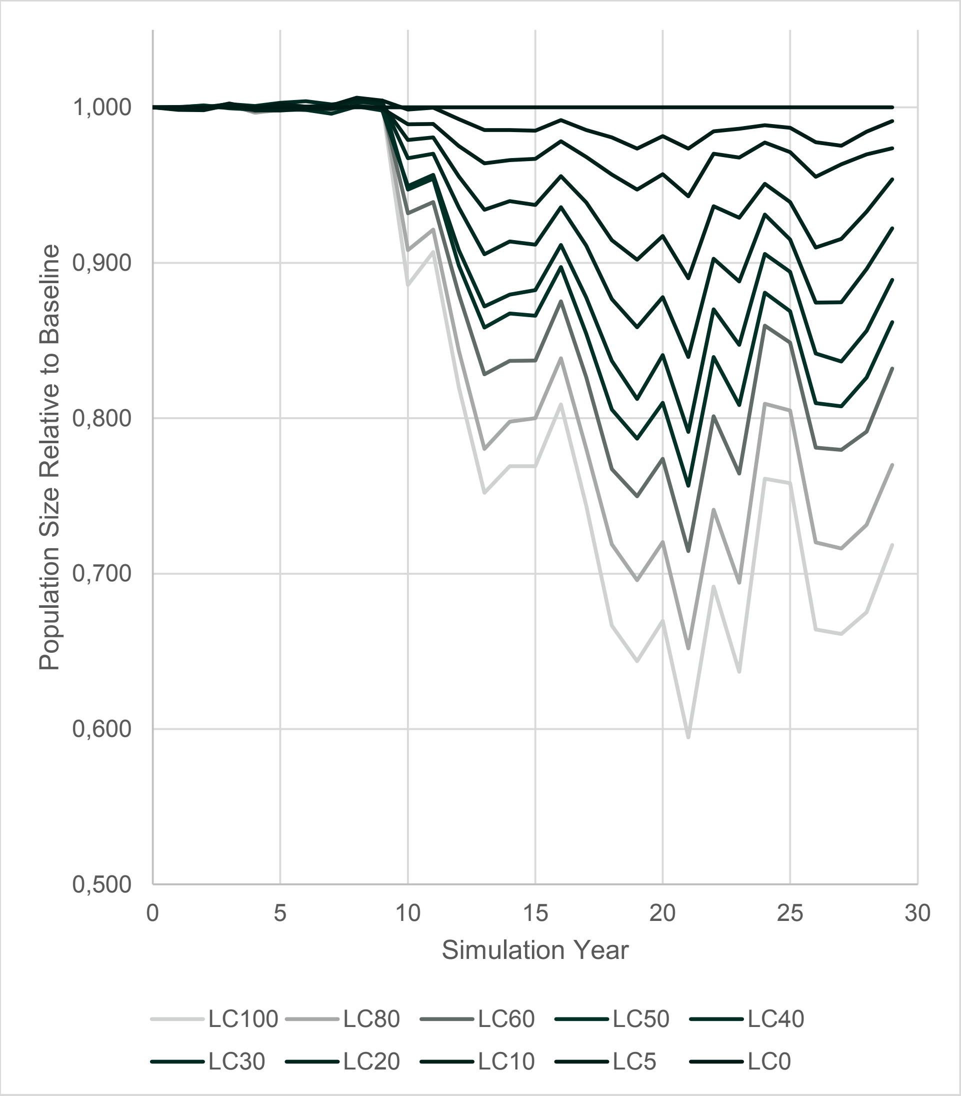


Figure 24: Time series of relative population size for LCx overspray scenarios and newt on Næstved landscape using weather from 1984-2014. Pesticide applied in year 10. Cropping scenarios is the standard WSO scenario.

Increasing LCx increased the impact on both abundance and occupancy in a close to linear pattern (Figure 25). However, the consequence of using 1950 or 1984 as the starting point for the weather input series was large. LC100 abundance scores under 1950s weather was -0.181 whereas using 1984-2014 weather the scores was -0.258, an increase in impact of over 40%. Occupancy impact also increased but by a much smaller percentage (5%).

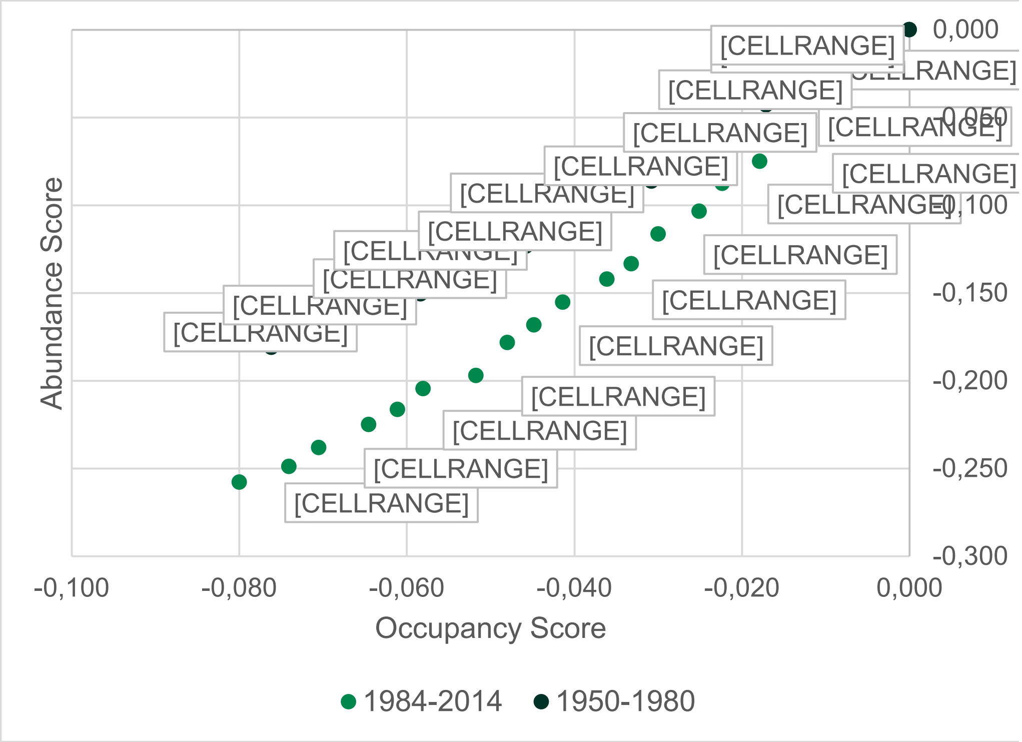
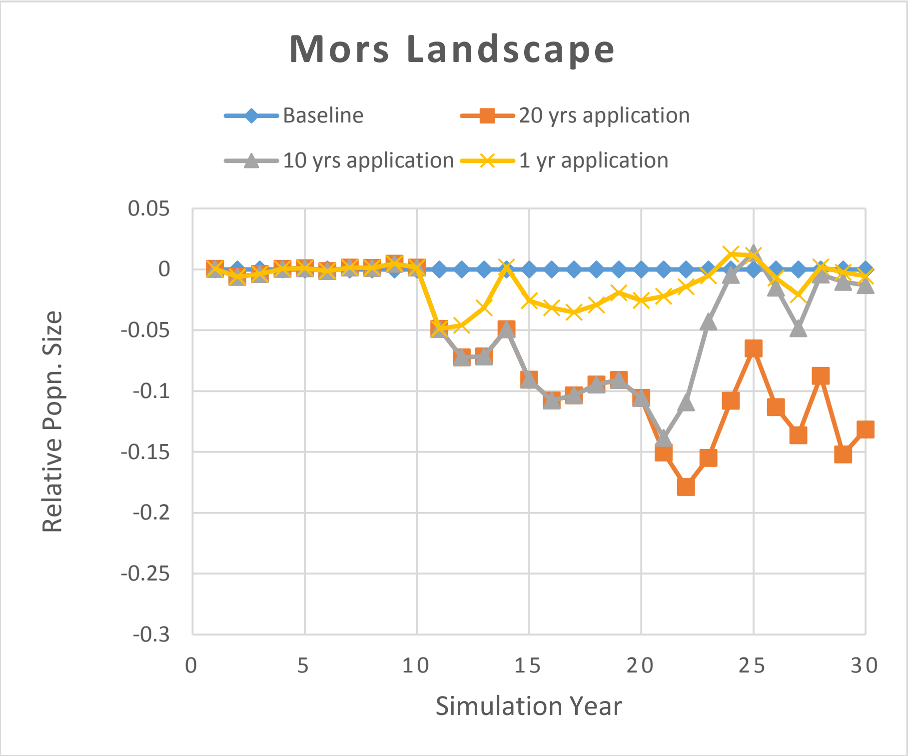


Figure 25: Changes in abundance and occupancy scores with increasing LCx for newts on Næstved under the WSO scenario and using two weather 30-yr patterns starting from 1950 and 1984.

The impact of year on year application is the most important endpoint for consideration, but it may be of interest to see how long impacts persist on cessation of pesticide use (recovery). The length of recovery time until the baseline and application scenario were indistinguishable was long. For one-year application it was 13-20 years depending how precisely the difference is measured (Figure 26). Note that partial recovery appeared to happen in Mors after 5 years, but this was only transient and not a full recovery. The pattern of recovery seemed similar for 10-year application, but the initial displacement from baseline was greater.



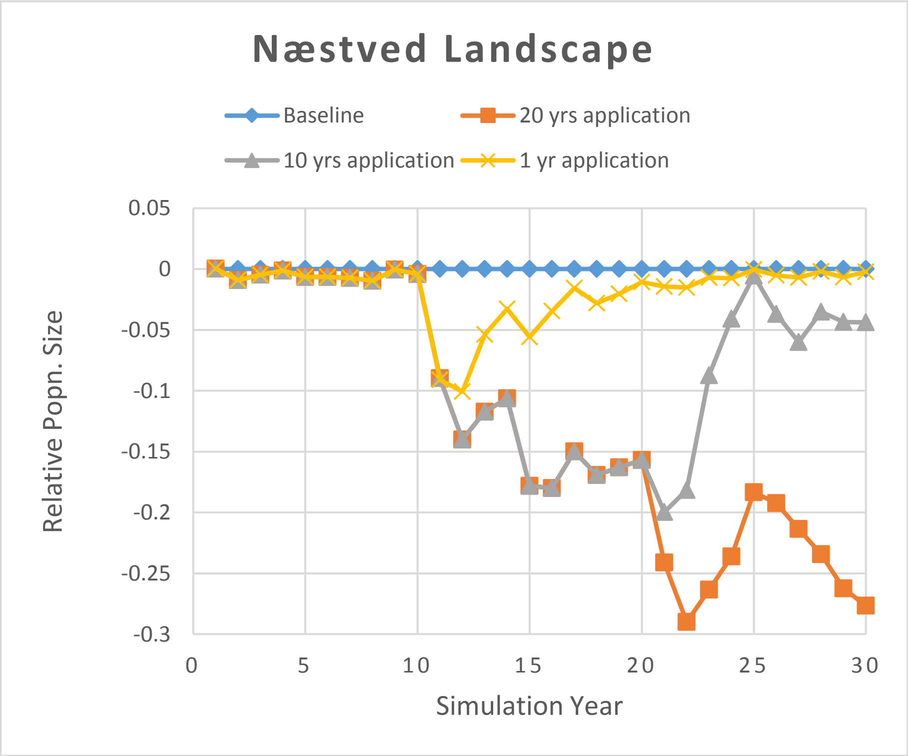


Figure 26: Time series of newt population size relative to baseline for overspray WSO LC100 scenario with application of pesticide for 1-year, 10-years and 20-years for Mors and Næstved landscapes. Pesticide application first occurs between years 10 and 11.

5.4.3 Stochastic pond quality scenarios

Assuming stochastic pond quality reduced population sizes by a mean of 75% (range 70.1-82.2, Table 30). Tønder landscape had the highest reduction and Esbjerg the least reduction, and there was no indication of a trend linked to baseline population size.

Landscape	Stochastic	Optimal	Difference
Es	20527	68734	70.1%
Hi	16312	57966	71.9%
Ka	10209	47171	78.4%
Ko	23198	91665	74.7%
Lo	22990	83631	72.5%
Mo	10267	39810	74.2%
Na	37670	139783	73.1%
Od	17061	71161	76.0%
To	15610	67214	76.8%
Tn	21676	121865	82.2%

Table 30: Baseline population sizes under stochastic and global optimal pond quality settings with the reduction in population size for stochastic ponds as a percentage.

Using stochastic pond quality settings between replicate variability was increased to a very high degree. Coefficient of variation (CV) for mean population impacts of 20 replicates of the standard WSOX10 scenario for global optimal pond quality was 0.01-0.03 across all 10 landscapes (Table 31), when compared to the mean baseline scenario for global optimal pond quality. For the first 20 replicates of the stochastic pond quality in the same scenario CV was 0.29-0.57 compared to the mean baseline conditions for stochastic pond quality. In some cases stochastic pond quality resulted in large increases in population size relative to mean baseline conditions (up to 158%) (Table 31). Examples of the variation shown by individual runs can be seen in Figure 27.

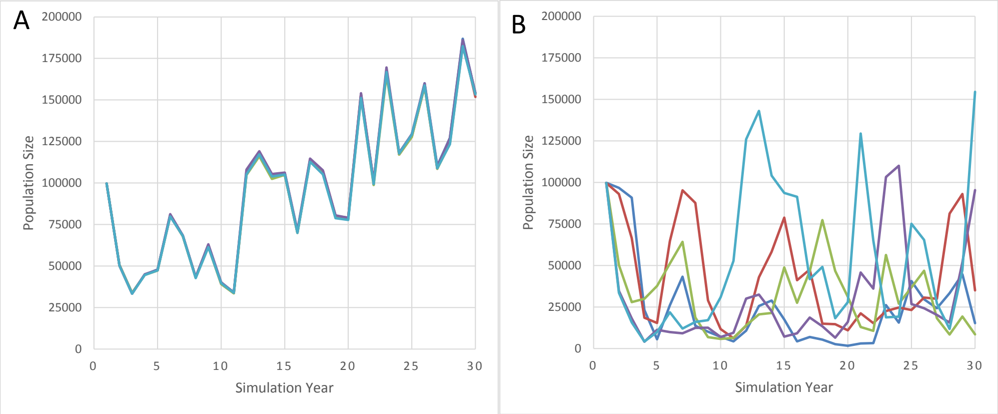


Figure 27: Five individual baseline runs for the newt baseline scenario on Næstved landscape assuming: A) global optimal pond quality; B) stochastic pond quality.

Due to the fact that the stochastic pond quality is a stochastic process operating at the beginning of each replicate run, it is not possible to compare replicate impacts directly. This would require fixing the variability of the ponds in a landscape and then running two scenarios, a baseline and pesticide scenario as pairs of replicates with the same pond conditions. This was not considered in the design of the model and therefore could not be done. However, the very large variation between replicates was most certainly driven by the distribution of pond qualities in the landscape.

	Optimal				Stochastic			
	Mean	**CV**	**Max**	**Min**	**Mean**	**CV**	**Max**	**Min**
Es	0.76	0.01	-0.26	-0.22	0.96	0.40	-0.55	0.98
Hi	0.79	0.01	-0.23	-0.20	0.91	0.45	-0.90	0.72
Ka	0.87	0.01	-0.15	-0.12	1.00	0.46	-0.84	1.58
Ko	0.47	0.02	-0.55	-0.51	0.54	0.53	-0.88	0.12
Lo	0.36	0.03	-0.65	-0.62	0.48	0.57	-0.86	0.40
Mo	0.56	0.03	-0.46	-0.39	0.71	0.48	-0.85	0.45
Na	0.35	0.02	-0.66	-0.63	0.87	0.42	-0.79	0.52
Od	0.60	0.02	-0.43	-0.38	0.60	0.50	-0.86	0.34
To	0.66	0.02	-0.36	-0.31	0.83	0.39	-0.61	0.46
Tn	0.71	0.01	-0.30	-0.28	0.90	0.29	-0.62	0.51

Table 31: Comparison of mean relative population impact, co-efficient of variation (CV) maximum impact and minimum impact amongst 20 replicates of the WSOX10 standard scenario using global optimal pond quality and stochastic pond quality settings.

Despite the high level of variation between replicates, impacts of the standard scenarios impacts on population size were highly correlated when based on 40 replicate runs (Table 32).

	Slope (b)	b +95% c.i.	b -95% c.i.	p-value
Es	1.000	1.301	0.698	6.08E-05
Hi	0.968	1.407	0.529	9.51E-04
Ka	1.129	1.431	0.827	2.53E-05
Ko	0.951	1.148	0.755	3.66E-06
Lo	0.914	1.218	0.611	1.19E-04
Mo	1.018	1.232	0.804	4.23E-06
Na	1.039	1.260	0.819	4.59E-06
Od	0.935	1.086	0.784	5.73E-07
To	1.040	1.241	0.839	2.24E-06
Tn	0.798	0.911	0.685	2.06E-07

Table 32: Regression slope, upper 95% confidence limit of slope, lower 95% confidence limit of slope and p-value for regression for all landscapes regressing the relative to baseline population level impact for stochastic pond quality against relative population impacts for optimal pond quality.

In all 10 landscapes the slope of the regression obtained by plotting the stochastic pond quality impact agains the optimal pond quality impact was within the range 0.798-1.129 (Table 32). Figure 28 shows an example from the Næstved and Esbjerg landscapes. In general where population impacts were smaller (e.g. Tønder landscape), fits between the two pond scenarios were poorer. This can also be seen in the greater variation at the lower end of effects in Figure 28. In all cases except Tønder, the 95% confidence limit on the slope of the regression included 1.0, suggesting that only in the case of Tønder were impacts significantly different between the two pond settngs. In this case impacts in all 10 standard scenarios were lower with the stochastic pond settings.

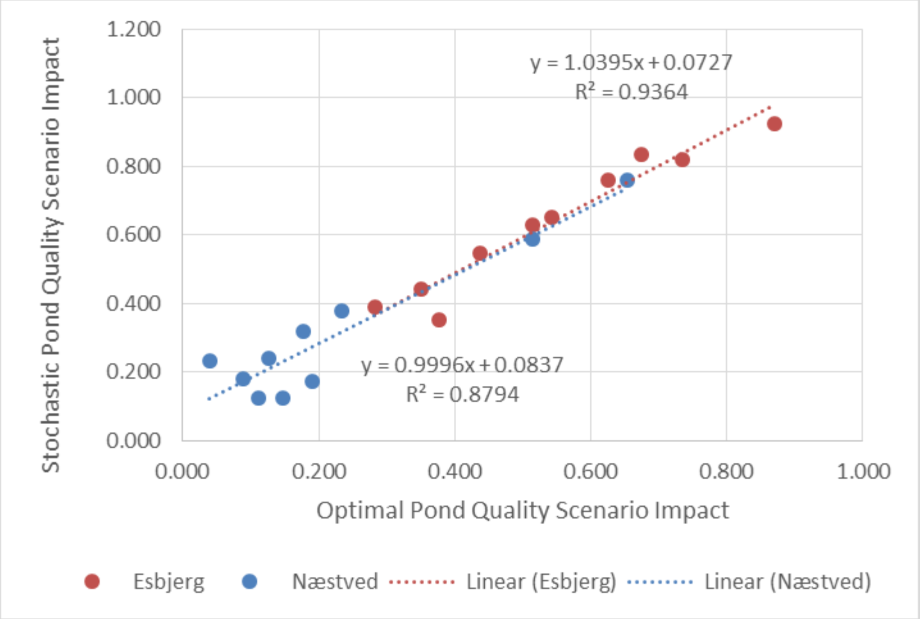


Figure 28: Relative impact of the nine standard scenarios resulting from assumptions of stochastic pond quality plotted against the impact assuming optimal pond quality on the Esbjerg and Næstved landscapes.

Re-running the LCx experiment using 1984-2014 weather and with 120 replicates resulted in a good correlation between stochastic and global pond effects. Here the impact with the stochastic pond effects setting was approximately 80% of that found under global optimal pond quality settings (Figure 29).

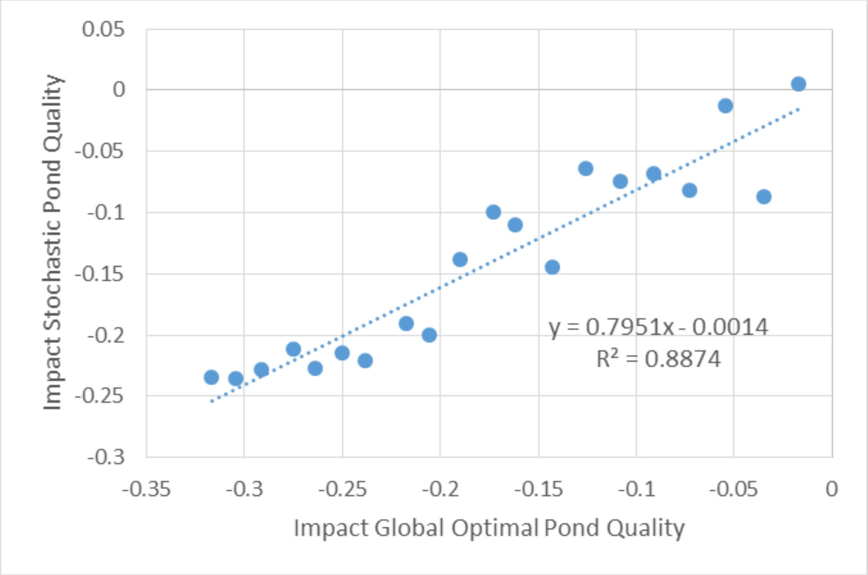


Figure 29: Impacts of LC_{00} to LC_{100} overspray scenarios using the stochastic pond quality setting plotted against the same scenario using the global optimal pond quality setting.

5.5 Combined analysis

General trends in impact on overall population size are difficult to find. Each species model reacted to the combination of scenario and landscape in its own particular way. Comparing relative impacts for beetle, skylark and newt to the hare population impacts showed no particular trend (Table 33).

Some trends were more apparent when comparing between landscape effects. For example, Tønder landscape was the landscape with the least impact for hares, beetles and skylarks in the WSOX10 scenario, however, it was also the landscape with greatest impact for skylarks and newts for the SBX10 scenario (Table 34). Of those landscapes with highest impact, 5 out of 10 landscapes were represented for the two scenarios (WSOX10 & SBX10) amongst all four species, suggesting that it is not possible to identify a generally worst-case landscape.

Baseline density was a factor determining the impact of the pesticide scenario. For example, the WSOX10 scenario produced a strong correlation between baseline density and impact on population size for beetle, skylark and newts under stochastic pond quality, but not for hares (Table 35). For beetles and skylarks, increasing baseline population size led to increased effects, but for the newts this trend was reversed. In the case of hares, the Lolland landscape was a clear outlier, and if removed from the analysis the result was a significant correlation (n=9, r= 0.691, p=0.039).

For SBX10 the pattern was similar for the newt with optimal pond quality, beetle and skylark, but for the hare, and newt with stochastic pond quality, there was no longer a significant relationship between baseline population size and effect of the pesticide (Table 35).

To exemplify the effect of long-term application, the impact of the pesticide in three standard scenarios (WSOX10, WSOX100 & SB100X10) were measured after one year, ten years mean (year 1-10 of spraying) and mean of 11-20 years of spraying after calculating mean effects from all ten landscapes (Table 36). In all cases, for all four species impacts in the first year were lower than the subsequent years, and impacts in the first decade of application were lower than the second decade of application. In most cases however, there was a much higher difference between year 1 and the first decade of application than between the first and second decade of application. Impacts on the beetle appear to be highest in the first year compared to the other species, with the hare not responding until after the first year.

		Esbjerg			Kolding			Næstved			Tønder	
Scenario	H/Be	H/Sk	H/Ne	H/Be	H/Sk	H/Ne	H/Be	H/Sk	H/Ne	H/Be	H/Sk	H/Ne
WSOx1	0.10	0.23	0.14	0.18	0.05	0.44	0.34	0.09	0.74	0.03	-0.01	0.30
WSOx10	-0.12	-0.11	-0.12	0.11	-0.28	0.42	0.09	-0.43	0.55	-0.05	-0.15	0.20
WSOx100	-0.31	-0.32	-0.27	-0.06	-0.49	0.22	-0.18	-0.69	0.22	-0.09	-0.23	0.12
WSOx1000	-0.53	-0.58	-0.53	-0.35	-0.71	-0.20	-0.51	-0.88	-0.36	-0.23	-0.38	-0.07
WSX10	-0.12	-0.15	-0.14	0.05	-0.32	0.33	0.03	-0.44	0.46	-0.04	-0.14	0.21
Sx10	-0.08	-0.17	-0.11	0.09	-0.24	0.16	0.18	-0.28	0.28	-0.02	-0.11	0.21
Wx10	-0.10	-0.11	-0.09	0.10	-0.27	0.28	0.16	-0.37	0.36	-0.01	-0.11	0.21
Ox10	-0.09	-0.17	-0.10	0.22	-0.13	0.19	0.19	-0.25	0.16	0.00	-0.09	0.20
SBX10	1.43	0.03	0.95	1.42	-0.24	1.86	1.87	-0.11	4.17	3.02	-0.18	6.44
SB100X10	80.44	-0.39	1.06	42.45	-0.44	2.63	162.02	-0.47	5.99	9.01	-0.44	8.95

Table 33: Population impacts of the standard pesticide scenarios relative to the impact on the hare population (positive values mean lower impact in hare, negative values mean higher impact in hare) for four landscapes.

	WSOX10				SBX10			
Species	Highest Impact		Lowest Impact		Highest Impact		Lowest Impact	
	Landscape	Impact	Landscape	Impact	Landscape	Impact	Landscape	Impact
Hare	Mors	0.48	Tønder	0.15	Mors	0.30	Odder	0.37
Beetle	Lolland	0.56	Tønder	0.11	Tønder	0.80	Karup	0.94
Skylark	Esbjerg	0.25	Tønder	0.00	Esbjerg	0.22	Tønder/ Lolland/ Næstved	0.03
Newt	Næstved	0.65	Karup	0.13	Tønder	0.89	Karup	0.40

Table 34: Highest and lowest impact landscapes and population reduction for the WSOX10 and SBX10 scenarios for all four species.

Scenario	Stats	Hare	Beetle	Skylark	Newt (optimal pq)	Newt (stochastic pq)
WSOX10	*r*	0.143	0.720	0.696	-0.527	-0.689
	b	3.02E-05	4.64E-06	6.49E-05	-2.97E-06	-1.481E-05
	p	0.694	0.019	0.025	0.118	0.027
SBX10	*r*	-0.022	0.818	0.760	-0.692	-0.560
	b	-1.91E-05	3.30E-06	6.18E-05	-3.44E-06	-1.06E-05
	p	0.951	0.004	0.011	0.026	0.092

Table 35: Regression statistics (correlation coefficient (r), slope (b) & p-value (p)) of population size impact of the WSOX10 and SBX10 scenarios regressed against the baseline density for all ten landscapes.

	Period	SB100X10	WSOX1000	WSOX10
Hare	Year 1	0.00	-0.02	-0.02
	Years 1-10	-0.22	-0.39	-0.18
	Years 11-20	-0.48	-0.71	-0.37
Beetle	Year 1	-0.44	-0.23	-0.21
	Years 1-10	-0.65	-0.48	-0.31
	Years 11-20	-0.70	-0.53	-0.34
Skylark	Year 1	0.04	0.01	0.01
	Years 1-10	-0.03	-0.07	-0.05
	Years 11-20	-0.07	-0.14	-0.10
Newt	Year 1	-0.49	-0.15	-0.12
	Years 1-10	-0.66	-0.37	-0.24
	Years 11-20	-0.82	-0.51	-0.37

Table 36: Relative population size measured at 1 year, 10 years and 20 years (the latter two 10-year means), for three scenarios and all species. Analysis based on mean landscape effects over all 10 landscapes.

With the exception of those scenarios with very low AOR-scores for either or both occupancy or abundance, there were between-species trends when comparing ratios of relative scores. Increasing toxicity generally increased the impact on occupancy more than abundance leading to lower A/O ratios (Table 37 WSOX1-WSOX1000). Increasing area treated had the same trend of increasing the occupancy effect more relative to abundance and largest effects on occupancy were found in the monoculture spring-barley scenario with forced three applications of pesticide (SB100X10). The exception was the skylark where the A/O ratio for WSOX1 seems low, probably due to the small impact of this scenario. However, impact of higher application rate scenarios X10-X1000 followed the expected trend.

	Hare	Beetle	Skylark	Newt
WSOx1	1.00	1.00	1.00	1.00
WSOx10	0.74	0.56	1.47	0.65
WSOx100	0.60	0.50	1.38	0.46
WSOx1000	0.45	0.47	1.23	0.35
Ox10	0.97	0.78	0.58	1.12
Sx10	0.82	0.68	0.70	0.94
Wx10	0.81	0.63	0.98	0.84
WSX10	0.74	0.56	1.50	0.68
WSOx10	0.74	0.56	1.47	0.65
SBX10	0.08	0.66	1.82	0.54
SB100X10	0.08	0.28	0.64	0.22

Table 37: The scenario mean of AOR abundance score divided by AOR Occupancy score for each species. Low responses < 1% are ignored in the analysis. The scores are standardised by the WSOX1 scenario for ease of comparison.

In terms of average change in relative population growth rate across all landscapes, there were clear differences between species. In the first decade of application, the rate of population decline in hares was highest in all standard crop scenarios except WSOX1 and OX10. In monoculture spring-barley, the newt population decline was higher than the hare, and generally high for all other scenarios. In the second decade of application all species except the newt showed continued declines relative to the baseline, although at a slower rate compared to the first decade of application (Table 38). Note here that even the skylark, which had very low impacts of the treatments, showed a slow long-term decline in 6 out of 10 scenarios.

	1st Decade Application				2nd Decade Application			
	Hare	Beetle	Skylark	Newt	Hare	Beetle	Skylark	Newt
WSOx1	0.000	-0.001	-0.010	-0.016	0.004	-0.002	-0.001	0.000
WSOx10	-0.029	-0.011	-0.013	-0.024	-0.012	-0.006	-0.002	0.000
WSOx100	-0.043	-0.017	-0.015	-0.030	-0.013	-0.004	-0.002	0.000
WSOx1000	-0.061	-0.025	-0.017	-0.032	-0.013	-0.003	-0.003	0.000
WSX10	-0.028	-0.011	-0.012	-0.024	-0.011	-0.002	-0.001	0.000
Sx10	-0.022	-0.012	-0.005	-0.021	-0.009	-0.005	0.000	0.000
Wx10	-0.023	-0.007	-0.008	-0.022	-0.010	-0.004	-0.001	0.000
Ox10	-0.015	-0.008	-0.002	-0.018	-0.005	-0.004	0.000	0.000
SB100X10	-0.018	-0.035	-0.008	-0.044	-0.007	-0.005	0.000	0.000
SB100X10b	-0.041	-0.008	-0.007	-0.036	-0.016	0.000	0.000	0.000

Table 38: Relative change in the population size compared for baseline for all species and averaged across all landscapes for each scenario for the first and second decade of pesticide application.

Another key dimension in ERA alongside toxicity is the scale of use. This is rarely considered explicitly but is a very likely factor in determining the impact of populations. This was investigated by correlating the impact on the population size with the scale of use of the pesticides in all comparable X10 application rate standard scenarios (WSOX10, WSX10, WX10, SX10, OX10, SBX10, & SB100X10). The area treated was calculated by multiplying the proportion of agricultural area covered by each treated crop for each landscape by the total agricultural area. This formed the *Area* treated. However, the different crops had different probabilities of treatment. Winter wheat had a mean application rate of 1.5 per year, spring barley 1.05 and winter oilseed rape of 0.33 (except spring barley SB100X10 which was 3.0). This was considered as a treatment index and a second explanatory measure was derived per landscape by multiplication with area for each crop (*AxT*). An overall correlation was also created for each species by pooling data across all landscapes. Resulting correlations are show in Table 39. Overall correlations ranged from 0.20 to -1.0. Only Lolland and the skylark gave a positive correlation coefficient.

Overall the expected negative correlations were good (-0.79-0.90) for beetle and newt, but very poor for skylark (-0.15 & -0.05), hare being intermediate. The *AxT* index was a consistently better performer for hares and in 8 out of 10 landscapes for beetles, and with one exception Area was better for skylark and newts. There was very large within species and measure variation in correlation in hares and skylarks (e.g. hare with area from -0.10 to -0.81 & skylark -0.17 to -0.89), but beetles and newts showed consistently high correlations between population impact and *Area* or *AxT* index.

Using the same approach the spatial distribution in terms of the occupancy index was also analysed (Table 40). The pattern was similar to population level effects, but with no clear pattern of impacts on skylark, and newt correlations were now better in all cases when using *AxT* index. Hare correlations were improved in those cases where they were poor for *Area* (e.g. Lolland from -0.20 to -0.96). Beetle correlations were also improved slightly, but now in all cases the *AxT* index was a better correlate than *Area*. There was no clear pattern of change in the overall correlation compared to the relative population effects across species.

Therefore, for three of the species assuming all other things are constant, the *AxT* index appeared to correlate well with spatial impact in terms of occupancy.

	Hare		Beetle		Skylark		Newt	
Landscape	Area	AxT	Area	AxT	Area	AxT.	Area	AxT
Es	-0.44	**-0.71**	-0.94	**-0.99**	**-0.59**	-0.45	-0.96	**-0.98**
Hi	-0.61	**-0.89**	-0.90	**-0.95**	**-0.68**	-0.37	**-0.95**	-0.91
Ka	-0.72	**-0.91**	-0.92	**-0.98**	**-0.44**	-0.26	**-0.95**	-0.95
Ko	-0.75	**-0.86**	-0.85	**-0.95**	**-0.81**	-0.50	**-0.96**	-0.88
Lo	-0.11	**-0.20**	-0.90	**-0.97**	**-0.17**	0.20	**-0.97**	-0.84
Mo	-0.51	**-0.70**	-0.84	**-0.97**	**-0.49**	-0.15	**-0.95**	-0.91
Na	-0.10	**-0.39**	-0.86	**-0.97**	**-0.49**	-0.06	**-0.97**	-0.86
Od	-0.22	**-0.42**	-0.86	**-0.95**	**-0.57**	-0.11	**-0.94**	-0.91
To	-0.49	**-0.60**	**-0.94**	-0.93	**-0.47**	-0.19	**-0.97**	-0.88
Tn	-0.81	**-1.00**	**-0.93**	-0.88	**-0.87**	-0.53	**-0.95**	-0.85
Overall	-0.48	**-0.62**	-0.87	**-0.90**	**-0.17**	-0.05	**-0.87**	-0.79

Table 39: Correlation coefficients for all standard X10 application rate scenarios population size reduction and Area = area treated, AxT = area treated multiplied by the average number of applications of pesticide for that scenario. Overall = correlation of pooled data across all landscapes. Bold indicates the highest correlation coefficient for each pair. df = 6 in all cases.

	Hare		Beetle		Skylark		Newt	
Landscape	Area	Area X Tr.	Area	Area X Tr.	Area	Area X Tr.	Area	Area X Tr.
Es	-0.71	**-0.93**	-0.86	**-0.99**	-0.41	-0.24	-0.88	**-0.99**
Hi	-0.78	**-0.98**	-0.83	**-0.99**	-0.45	-0.09	-0.88	**-0.97**
Ka	-0.63	**-0.89**	-0.83	**-1.00**	-0.05	0.07	-0.87	**-0.99**
Ko	-0.88	**-0.98**	-0.80	**-0.98**	-0.41	0.07	-0.80	**-0.98**
Lo	-0.89	**-0.96**	-0.84	**-1.00**	0.69	0.97	-0.83	**-0.99**
Mo	-0.61	**-0.78**	-0.79	**-0.99**	-0.24	0.18	-0.85	**-0.98**
Na	-0.36	**-0.62**	-0.81	**-0.99**	0.37	0.75	-0.75	**-0.98**
Od	-0.47	**-0.69**	-0.80	**-0.98**	0.32	0.77	-0.80	**-0.98**
To	-0.56	**-0.66**	-0.90	**-0.97**	-0.14	0.16	-0.86	**-0.98**
Tn	-0.84	**-0.99**	-0.92	**-0.93**	-0.70	-0.28	-0.82	**-0.99**
Overall	-0.52	**-0.71**	-0.81	**-0.95**	-0.02	0.13	-0.79	**-0.95**

Table 40: Correlation coefficients (r) for all standard X10 application rate scenarios occupancy impact and Area = area treated, Area X Tr. = Area Treated multiplied by the average number of applications of pesticide for that scenario. Overall = correlation of pooled data across all landscapes. Bold indicates the highest *r* for each pair except skylark where no clear trends were obvious. *df* = 6 in all cases.

6. Discussion

Landscape-scale population-level ERA is a new concept in regulatory risk assessment, but is gaining ground quickly, not least due to new EFSA policy for moving towards landscape-scale for all risk assessment. But because it is new, this type of assessment brings a wide range of new issues that need to be considered. Therefore, although the scenarios presented here require further work to examine many details of the ERA, they do provide a broad picture of some of the issues that need to be addressed.

The simulations run in this study were designed to answer two main questions regarding landscape-scale population-level:

- What is the role of landscape context when carrying out the ERA?
- To what extent do species ecology and behaviour influence the impact predicted in the ERA?

6.1 The role of landscape context

There was no case in the data presented here where effects were found, and where landscape context did not alter the outcome of the risk assessment. However, there are different facets to this. The first being that different landscapes have different carrying capacities for the different species. Baseline densities for the hare varied by factor 16, whilst for the other species baseline densities varied by factor 2 (beetle) and 3.5 (skylark and newt). In the case of the newt the proportional variation was the same whether stochastic pond quality was used or not. Highest densities for skylark and hare were in Lolland, whilst the beetle was most abundant in Himmerland and the newt in Næstved. Lowest densities were found in Karup for hare and skylark, but in Lolland for the beetle, and Mors for the newt with optimal pond quality, and Karup and Mors joint last for the newt with stochastic pond quality.

In all cases, population size in WSO scenarios was somewhat related to the impact of the pesticide in proportional terms. This means that in the case of the hare, skylark and beetle the higher the carrying capacity of the population the higher the population impact of the pesticide, but for the newts this trend was reversed. For the hares there was one clear outlier - the Lolland landscape (with extremely high density of hares), resulting in a lower impact of the WSO scenarios. This outlier can be explained by the very high uniform distribution of hares in the Lolland scenario, and the relatively low area occupied by treated crops in the WSO scenario.

This might suggest that if the area is more uniformly treated then the effect of the pesticide would be more clearly related to population size. However, this does not seem to be the case. The spring barley monoculture scenarios did not show a better correlation between impact and population density.

Correlations between population effects and baseline density from the SBX10 scenario (where all fields can potentially be treated) showed more pronounced differences between species. Impacts on beetle populations were very strongly positively correlated to baseline population size, skylarks were also strongly correlated, but there was no correlation for hares. Newts were negatively correlated, although only significantly under global optimal pond quality. This suggests that different factors were operating, bringing the animals into contact with the pesticide to a larger or small degree, but also that population processes were important.

In the case of the beetle, high population densities are related to large areas of good habitat, in this case arable fields. When all these fields are treated the population is likely to be uniformly affected and will be effectively excluded from large areas due to its slow dispersal. The case of the skylark is similar, but here vacant areas will be repopulated by birds from the small areas of suitable breeding that are not treated. However, movement is not completely free since skylarks are territorial and will therefore move in to vacant area, but will not move out if they have established territories. For hares, the short-term home-range is much larger than a single field, and hares move freely within the landscape (unlike beetles and skylarks). Therefore, under these scenarios population impact is determined more by factors that are directly correlated with population size as a function of treated field area. These include the toxicology of the pesticide, and the relative importance of in-field vs off-field habitat to the hare, which will be landscape dependent. It is easier to see why newt populations consistently show a reversed trend between population size and pesticide impacts. In this case newts are dependent upon ponds which are fixed in the landscape and the pesticide affects the newts when in their terrestrial stage around the ponds. There is no attractant or repellent function of the fields or treated crops, hence the processes in play are related to the newts ability to withstand perturbations. Higher newt baseline densities will mainly be a function of higher numbers of ponds, and higher numbers of ponds will increase the chance of recolonization, as well as high densities increasing the chance of survival for enough newts to facilitate local recovery by reproduction. Large newt populations will therefore be more robust to perturbation than small ones. This is discussed further below under newt species ecology.

The case of Lolland and the hare also indicates another main factor in determining the impact of the pesticide; that is the scale of use as determined by the farming practice carried out in the landscapes. Treated area was a good predictor of impact on population size for beetles, skylarks and newts, but not hares. However, if combined with a treatment index, taking account of the number of treatments per crop as well as area, then beetle predictions were improved and impacts on hare populations correlated in general well with the index. For occupancy measures the combined area X treatment index worked well for all species except skylark. Within skylark good correlations between the area X treatment and impact were found only in Lolland, Toftlund and Odder landscapes. There appears to be no obvious reason for these three landscapes to be different from the other seven. Across all species and landscapes, similar to baseline population size, the impact of area treated and/or treatment index was generally correlated with impact, but was variable between landscapes and species.

Overall, landscape effects differed between species but caused variation ranging typically from 30-100% increase from the best-case landscape to worst. In the case of the hare it could be concluded that farming was approximately twice as important than landscape structure, but this conclusion is unlikely to be valid for the other species. In particular, the newt model includes little that directly interacts with farming activities or crop types, but is heavily dependent on the distribution and number of ponds; therefore, landscape structure becomes much more important.

6.2 Species ecology and behaviour effects on ERA

6.2.1 Hare

The hare model represents a wide ranging species with potential for multiple exposures throughout its breeding cycle. ERA effects on this species were strongly linked to landscape structure and farming. There are a number of reasons for this. The first being that the movements of the hares occur over a large area and therefore the exposure potential is high, but this is reduced if movement activity is reduced. Reduction of movement happens for two contrasting reasons in the model, either because local conditions are very good and dispersal is not needed to obtain resources, or because conditions are generally poor and resources are locally distributed. The second link to landscape is that the hare is very dependent on vegetation structure for movement through the landscape, and for resources. Vegetation structure is profoundly influenced by the farming activities determine crop coverage and crop management, and these actions can also directly impact hares (e.g. mortality of leverets caused by vegetation cutting).

The hare was also typically predicted to follow long-term declines under most of the pesticide scenarios. This is another consequence of the mobility of the species. Since the hares have no fixed territory, any hare is likely to come into contact with a generally applied pesticide at some point. Therefore, reduction in reproduction rates will occur to a greater extent as a whole population effect than in the spatially restricted species such as newt and beetle, together with the long life-span this results in longer term declines.

6.2.2 Beetle

The beetle differs from the other species considered because it is in the field at the time an insecticide is assumed to be applied, and as such, significant local impacts are expected. What is important in this case is whether the population will suffer long-term depression or declines.

The beetle also differs from the other species in its life-cycle length. Both the life-cycle and the fact that beetles are generally in the field at the time of spraying affects the impact of the ERA. Impacts of the pesticide on beetles was initially higher than for any of the other species. This is partly due to the direct action of the pesticide in causing direct mortality of the adults, but also to the fact that there is no time delay between impact on the adults and the next generation, unlike all the other species simulated. Previous beetle simulations suggest that dispersal is limiting recovery (Topping, Kjaer et al. 2014, Topping, Craig et al. 2015) in this species, but local recovery is fast as long as the beetle is not extirpated from an area.

We used the beetle to investigate what happened if ERA was carried out for larval and adult life-stages separately, and then combined. The question of how this should be done seems to be open. Combining the ERAs for two life-stages would be combine the risks additively, assuming these are independent. However, the assumption of additive effects of pesticides on different life-stages is probably conservative in the cases like the beetle where there is a strong density-dependent control. In all landscapes, impacts of the combined ALX10 scenario were lower than summed effect of the two scenarios run separately (2.5% and 22.4% lower). The size of the difference was dependent on the landscape (Figure 14), and was not proportionally constant. This indicates that not only do we have a different result of the ERA by combining impacts in a single simulation, but that there is an interaction with landscape that accounts for an order of magnitude difference in this effect.

This phenomenon is another aspect of the doomed surplus concept in ecology (Errington 1946), whereby predation has a lower impact than expected because of density-dependent compensation. This means some proportion of the population will die anyway, and if up to that proportion is predated it should have no overall population influence. The extent to which this will happen in an ERA will depend on the speed of the life-cycle and the life-stage affected. Fast life-cycles will tend to compensate faster than slow, and killing animals in the stage where density-dependent mortality acts will have a lower effect than killing them in later stages. This was seen here between adult and larval beetles, where larval impacts were much lower than adults even with an assumed 10X higher sensitivity to the pesticide.

6.2.3 Skylark

Skylark simulations indicates the importance of timing of effects relative to life-history. Standard scenarios had very low impacts because the most critical breeding time is before the standard scenario effects. Despite this, a clear impact of the pesticide was shown, even if at a lower level. This demonstrates the importance of the timing, which was confirmed by the eggshell thinning scenarios with spring-barley spraying moved 1 month earlier. After the fact, this seems obvious; nevertheless, this underlines the importance of including realistic Good Agricultural Practice (GAP) in the risk assessment. Note that this also means that if there are multiple GAPs with different timing of application then evaluation of all of these is necessary.

The eggshell scenarios exemplify some of the interesting details of ecology that the models can provide. In Eggshell Scenario 4 (advanced spray timing), assumption of independent egg effects were worse than clutch effects. The cause of this is that in an early spraying situation loss of a whole clutch will trigger a re-laying, hence to a degree compensation is possible. The model assumed that even for highly persistent compounds the probability of clutch loss was the same as before, i.e. is independent of the first loss. Therefore, under these circumstances clutch replacement is viable and probability of survival of one clutch is higher (e.g. 50% loss of first attempt followed by 50% loss of the second attempt leads to an overall 75% loss). Single egg loss is not replaced and so the rest of the brood is reared, preventing re-laying. In Eggshell Scenario 3 (no advancement of spray date), clutch loss impacts were greater than egg loss because there was no chance of relaying due to late spray. With application at this late point, the loss of some eggs was slightly compensated for by better survival of the smaller broods. However, in all cases in Eggshell Scenario 3, very few clutches were affected.

6.2.4 Newt

The great crested newt model represents a species with limited dispersal but unlike the beetle, it is reliant on off-crop habitats for breeding. The results of the newt simulations raise a number of issues related to its ecology. One of these aspects is the low dispersal ability and therefore low landscape permeability leading to long recovery times. This will be a general feature of animals with low dispersal ability in situations where extirpation from an area can occur. The degree and pattern of recovery will however be different in each case. In the case of the newt the strong effect of weather creates a situation where in overall population recovery appears to take place but the effect is only temporary. This is linked to probably the most significant aspect of the newt ecology pertaining to risk assessment, and that is the huge year-to-year variability in population size. Even under standard conditions of optimal pond quality the newt population model created large year to year fluctuations in a multi-year cycle which must be driven by the weather but did not replicate weather patterns directly. These patterns match real newt population dynamics well (as far as there is data to judge this). If we accept this as a natural feature of the population then it should be no surprise that newts primarily showed impacts in occupancy and not abundance. Since local recovery from low population levels seems to be a natural part of the population dynamics, a local all-or-nothing response seems reasonable under any severe stress.

The effect of introducing stochasticity in pond quality was primarily to increase between replicate variance to a very large degree, but few other effects were noted. The way this stochasticity was implemented was that for any single replicate pond quality was fixed in space but varied between replicates. The resulting high variance indicates that the actual pattern of pond quality will be very important in determining the impact of the pesticides through two mechanisms:

1) Changing the proportion of the population exposed. This will be important if in reality pond quality is correlated with other landscape features e.g. fields;
2) Recovery as a result of local dispersal. If pond quality is associated with e.g. off-field ponds, then potentially there can be an interaction between pond quality and recovery potential for the poorer in-field populations.

This means that in reality the coincidence between high pond quality and pesticide use will have a large impact on the ERA. Using optimal pond quality in this case created a more conservative assessment since impacts were always higher than with stochastic pond quality (although similar and well correlated). However, as with all other factors investigated in this study there was considerable landscape to landscape variation in the degree of predictability of the stochastic pond effect from the optimal pond quality effect. For real ERA the recommendation based on this model is to use optimal pond quality setting for the assessment unless actual pond quality can be predicted from maps and incorporated. This might be possible since pond quality is linked to pH, surrounding vegetation and the presence of fish. Of these, the first two are visible in GIS mapping data available, and could therefore be incorporated. Presence of fish might be based partially on pond size (and potential to dry out), and pond age. Further work might be possible to establish a workable model to improve the ERA.

As a more general point for this type of mechanistic simulation model, the incorporation of pond quality stochasticity highlights one of the problems of using stochasticity to cover unknown mechanisms, it simply adds noise but little in the way of understanding.

Newt overspray scenarios were used to illustrate translation of laboratory toxicity to population-level effects. However, these also illustrated an important potential effect in that at low lethal concentrations there was no observable effect on population size for the first few years. Although subsequent effects were relatively small, about 2.5% population impact, depending on the specific protection goal considered these may be important. Hidden small but long-term effects may be a feature of many systems, and would not be noticed using other methods either because of the short term (field studies) or difficulties with the limit of detection for small effects (Brock, Hammers-Wirtz et al. 2015).

6.2.5 Suggested improvements to the newt model

The newt model as used here is able to evaluate impacts of overspray and exposure to environmental concentrations of pesticide in the terrestrial environment. Whilst this is fine as far as it goes a major improvement would be to integrate the aquatic ERA with the terrestrial ERA. The newt model has been set up to be able to respond the aquatic concentrations of pesticides if these can be simulated; this is not easy, however. The major obstacle is not in the newt model itself but in the landscape handling of pesticides and the lack of suitable exposure models for ponds. The standard way to do this is to use the FOCUS scenarios (FOCUS 2001). However, these are static scenarios that are not suitable for integration into the dynamic model, not least because they define conditions that a part of the model dynamics or landscape inputs, and therefore variable. Consequently, a new pond exposure module will need to be developed. This will probably need to incorporate drift, run-off, in and out-flow, and concentration effects due to drying out or rainfall.

Other important developments would include the development of a mechanistic basis to derive pond quality from, including vegetation structure around the pond, soil type and age. This will potentially alter the distribution of the newts, and therefore their exposure, based on landscape characteristics. If handled appropriately this need not increase stochasticity of model outputs between replicates, although it almost certainly will contribute to greater between landscape variability.

In terms of representing the newt biology and ecology the main improvement will probably be the development of a more detailed model for newt development. This is likely to further increase the impact of weather since temperature will be an important driver, but also may alter the risk assessment by altering timing of newt activity and therefore exposure.

Overall, collection of data targeted at evaluating the newt model performance is suggested. Suitable literature data was not extensive, although much may lie hidden in ‘grey literature’. Therefore, data collection to support the model evaluation is important and can determine the extent to which biological process, but most importantly mortality are presented in the model. Given the newt model’s heavy reliance on weather as a driver, these relationships should be tested under conditions not present in the original study by Griffiths Sewell et al (2010).

6.3 Endpoints

Endpoints for the population ERA are the measures made to assessment impact of the pesticide on the landscape-scale population of the animal simulated. These endpoints are entirely new to the risk assessment since the concept of population modelling and landscape-scale

ERA is new to the regulatory risk assessment.

6.3.1 Impacts

To measure impact we need a baseline situation to compare population effects against. This is termed the no pesticide situation and represents a model identical in all respects to the scenario used to test a pesticide except that the particular pesticide under evaluation is not applied (but see The Regulatory Scenario 4.4). The baseline population size will fluctuate in time, and this defines the normal operating range of the population. To create the baseline, the model needs to be carefully tested to determine whether it performs closely to the real world. Examples of this can be seen for partridges, hares, voles and skylarks (Topping, Hoye et al. 2010, Topping, Hoye et al. 2010, Topping, Dalkvist et al. 2012, Topping, Odderskaer et al. 2013), as well as the newt model testing (section 6.5).

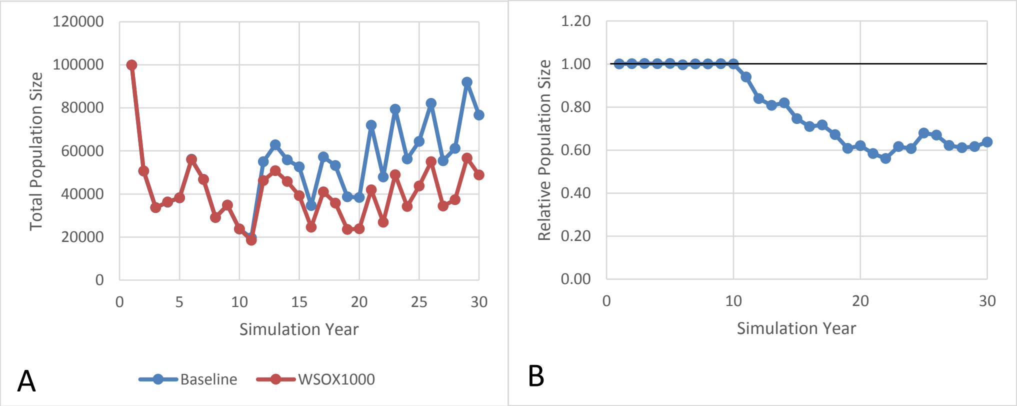


Figure 30: A) Newt baseline and WSOX1000 total population size on Esbjerg landscape. B) The same baseline and WSOX1000 scenario data presented as relative population size.

Once we have a baseline, addition of the pesticide to the otherwise unchanged scenario produces an altered population curve. Note that the basic shape of the curve is the same as the baseline, but the height of the curve is different (Figure 30A). This is because the other main drivers of population size e.g. farming, landscape and weather are identical between the scenarios. However, comparison of the raw numbers between runs is difficult and the population size relative to the baseline is used to facilitate easy comparison (Topping, Dalkvist et al. 2009) (Figure 30B). This allows comparison of the baseline and pesticide scenario directly and is the method used in this report.

The overall population impact in terms of population size is the simplest endpoint available. However, as introduced by EFSA Panel on Plant Protection Products and their Residues (PPR) (2015), for non-target arthropods another endpoint can be considered. This is based on the AOR-index (Hoye, Skov et al. 2012). This endpoint is more detailed and describes the change in abundance (population density where the population occurs), and occupancy (the relative proportion of the landscape occupied by the population). As with overall population impact these measures are relative to a baseline (e.g. Figure 25 shows an example of changes in occupancy and abundance for increasing LCx overspray scenarios in the Næstved landscape). The effect of increasing toxicity is clearly seen both in terms of changing newt population, abundance and newt distribution, which in this case shows a close to linear response. Multiplication of the abundance and occupancy raw scores provides an estimate of population size, which can then be compared using the AOR scores. The use of occupancy as an endpoint increases the descriptive power of the ERA. Adding the spatial component permits the evaluation of local impacts, for instance if it is important that range is not compromised (e.g. if we want in-field predator control), then occupancy might be considered a sensitive endpoint; this would then need to be defined in the specific protection goal (see below).

Probably the most controversial impact endpoint introduced by this study is the use of relative population growth rate (pgr). Pgr has however, been suggested as an important endpoint in risk assessment before (Pery, Mons et al. 2004, Hanson and Stark 2011). However, it has also been criticised as being less responsive compared to population size (Wang and Grimm 2010). Currently, the main point of using pgr in a risk assessment has been to evaluate the risk of extinction (Pery, Mons et al. 2004, Wang and Grimm 2010, Hanson and Stark 2011). However, in this report we suggest the use of pgr as a descriptor of the rate of change of population size over a particular period, and as a relative measure compared to the baseline pgr. This differs from the previous uses in three ways:

1) The measure is taken as a relative measure allowing differences in pgr for both increasing and decreasing populations to be more easily combined;
2) A pgr of >1 does not automatically confer no risk, since it is not whether the population is at risk of extinction that is the issue, it is the impact of the pesticide on the resilience of the population;
3) We suggest the use of pgr over a limited period of the simulation to identify long-term population trends, which may lead to significant long-term effects even when short-term impacts seem negligible.

6.3.1.1 Year-on-year effects

Impact can be measured as single pulse or as the result of year on year application. In the case a single year impact of the LC100 overspray scenario for the newt a 9 % reduction in population size occurred in the first year, but continuous use led to a 20% population decline after 10 years. It is clear that an effect of year-one-year application of the pesticide occurs that is greater than the single year effect. One potential explanation of this in e.g. WSO scenarios is that the pesticide is not applied to the whole landscape at once, thus it may take time to get complete landscape exposure. However, this cannot be the whole story. If this were the only mechanism in play then the monoculture spring barley scenario with all crops sprayed every year (SB100X10) would have a large impact in the first year but no greater impact in the following years. This was not the case as is clearly seen from the analysis of 1, 10 and 20 year effects (Table 36) This indicates that population processes, not just the spatially changing distribution of pesticide are a major factor in this long-term decline.

A further reason for needing to look at long-term population effects is that depending upon the size of acceptable impact, it may difficult to identify a population response from a single year of application. Figure 24 shows that for newt overspray scenarios and LC_5 it took 3 years before the population impact was observable in the model, but after that period the effect was clear averaging 2.5% and never returning to the baseline. An effect such as this will be missed in both short-term simulation and also in any short-term field experiment.

6.3.2 Recovery

If we are comparing annual population status using the impact endpoints described above then there is no need to consider within season recovery, since if this does not occur there will be population impacts. However, if population impacts are allowed recovery should be possible. This situation might occur if a pesticide were allowed only for a short period of time, accepting population level effects but with the expectation of population recovery after cessation of pesticide use.

Like impact, assessing recovery can be done by comparing changes relative to a baseline condition. This example provided in this report was based on a 100% mortality overspray scenario for the Næstved landscape and the newt model. In this example (Figure 26) recovery seems to occur in the 10-year application scenario by year 25, but full recovery does not actually take place in the 20 years following cessation of pesticide application. Even after one year's application, recovery takes 15 years in this system. After 10 years of application and 10 years recovery the population was still at 95% of its original size. This indicates a serious problem with landscape-scale spatial recovery. Similar slow recovery from spatial perturbation has been shown for the ALMaSS vole model previously (Dalkvist, Sibly et al. 2013), and is linked to the dispersal ability of the organism. This in turn is affected by the landscape structure altering landscape permeability, which has been shown to be important in terms of recovery in spatially distributed aquatic insect populations (Galic, Hengeveld et al. 2013). This is not surprising since landscape permeability is known to be a critical factor in long-term survival of spatially distributed populations of conservation interest from insects to large mammals (Singleton, Gaines et al. 2004, Powney, Broaders et al. 2012).

6.3.3 General endpoint remarks

For all endpoints, the period over which they are measured is important. For impacts, these should be measured over a period that is long enough to avoid bias due to local temporal effects caused e.g. by weather, and should also avoid initial years of treatment since effects take time to build up. Similarly, for relative pgr, it is important that this endpoint is measured after a period over which it is reasonable to expect the population to have equilibrated. Here we used 10 years, but this is an arbitrary choice, and shorter or longer time-scales may be justified. Note that for all cases 5-10 years were needed for the initial population decline to slow.

6.4 Specific Protection Goals (SPG)

For populations at landscape scales the main features of interest will be the abundance of the animals, the distribution of the animals and potentially their condition (e.g. size, reproductive state). However, condition is a difficult thing to generalize across all species, and hence it is likely that as done in this report abundance and distribution are the primary focus. Therefore, when defining SPGs, it is important to consider the dynamics of populations in nature such as changes in abundance and distribution over time. Distribution and abundance are rarely, if ever, static. How, then, may SPGs involving distribution and abundance be looked at more dynamically and realistically? The answer is of course to use population models such as is done here. However, this allows us to simulate the population processes and impact of the pesticide, and if we have the SPGs defined in these terms we can compare effects to these and determine whether there is concern or not, but more often than not, the problem is definition of the SPGs in the first place. An example of how this can go wrong is the current EFSA bee guidance (European Food Safety 2013). This guidance sets the SPG threshold for effects on forager mortality at 7% of the colony size. On the face of it, this is fine and there was a defendable basis for the decision based on discussion with stakeholders. However, the 7% threshold appears to be unworkable in practice because it is below the minimum detectable difference possible with the power of most field studies (Bakker 2016). Here, simulation provides another opportunity. That is taking laboratory effects e.g. a dose-response mortality curve, or an egg-shell cracking threshold, and evaluating the biological relevance in terms of the population-level landscape scale endpoints. The potential to use simulation to determine biological impacts provides risk managers with a much more flexible tool for decision making. Figures such as the LCx figure for the newt (Figure 25) can be constructed and laboratory endpoints translated to population endpoints via suitable regulatory scenarios. This allows the risk manager to work with e.g. abundance and occupancy changes rather than the more abstract LCx or ECx.

Once defined SPGs can then be used to set the limits for ERAs for new pesticides and scenarios not used in the creation of the SPG.

6.5 The Regulatory Scenario

Typically, in risk assessment, the concept of a realistic worst case is used to define a situation that is not extreme, but realistically conservative. In terms of simulation models, this concept is applied to what can be termed the regulatory scenario (EFSA Panel on Plant Protection Products and their Residues (PPR), 2014) (Products and their 2014). This is the scenario used to carry out the ERA and assess the impact relative to the SPG; and of course requires a pre-defined landscape context, as well as the toxicological information and use information for the pesticide. This context will therefore need to include the weather, landscape structure and farming.

6.5.1 Landscape and farming

It is clear from the variability in terms of landscape effects on the ERA that pre-defining a single realistic worst case is not easy. Within a species there was some consistency between scenarios. For example, the WSOX1-X1000 series of scenarios are very similar only varying in toxicity of the pesticide, and thus the worst-case landscape was consistent. However, different scenarios can also change the landscape that is worst case. When changing from WSO set of scenarios to the monoculture spring barley scenarios the worst case landscape changed in three out of four species.

Similarly to landscapes, farming realistic worst-case is not always easy to define. In this study, we have used two main types of farming, the actual farming that is carried out in a landscape, and the assumption of mono-culture spring barley. In two out of four species (beetle and newt) the mono-culture spring barley (SBX10) was more worst case than realistic farming at the same level of toxic effect (WSOX10); but this means that in 50% of the cases the monoculture and therefore larger scale of use did not make the situation worse.

6.5.2 Weather

The weather has a pronounced effect on all species modelled, but this effect is different between species. Therefore, like landscape a worst-case scenario is not easy to find as a general case, and even if worst-case baseline weather were to be identified this does not mean it would be worst case for the ERA. So the choice of weather needs to be made on a pragmatic basis. For ALMaSS simulations we have typically used two types of weather data. The first is a short time series e.g. 10 years which is repeated (e.g. Topping, Sibly et al. 2005, Dalkvist, Topping et al. 2009). This has the advantage that the weather signature can be identified as a recurring pattern in the population size curves with time (once enough replicates are created). The pesticide phase can be directly compared to the same conditions as the unsprayed baseline phase and long-term population changes driven by weather will not occur. The second way is the used in this study, which is to use a long-term weather data set. This has the advantage that it can represent a long time period working backwards from the current situation. It can also be argued that the longer-term data set is more likely to include future conditions than a shorter data set. However, this may not be the case if directional climate change occurs. Using either repeating or long-term weather for simulation therefore requires careful consideration, and clear statements about what has been used. In the case of the newt, evaluation of impact of the LCx scenarios on occupancy could be altered by approximately 30% by selecting an earlier weather data set. Given the direction of effect there is clearly no justification for selecting the earlier data set. Conversely, if the impact of the older weather data had been 30% greater rather than less, it could be argued that the older data represents a realist worst-case situation that may occur in the future, and therefore be selected. In all cases, unless explicitly evaluated these effects could go unnoticed.

6.5.3 Baseline scenario

In this report, we have used the term baseline to mean the same as the treatment scenario but without the pesticide. However, this is a simplistic view. Removal of the pesticide from a treatment scenario only makes sense from a theoretical experimental view. In real-life, this will not happen in this way. If for example, an insecticide is being assessed, then it is quite likely that this is a replacement product for another compound. Hence, the baseline for the risk assessment, if you take away the pesticide is no longer realistic. Other effects might also manifest themselves. By removing the pesticide in the simulation the farmer no longer drives his tractor onto the field for that operation. For some species, this can also have an impact. In skylark simulations, the effect of removing spray applications was found to be negative due to the lack of suitable structure in mature cereal crops for successful breeding if tramlines were not kept open (Topping and Odderskaer 2004, Topping 2005).

It might also be considered whether assessing the effect of the pesticide in isolation from all other stressors is the best way forward. This has the advantage that it is a simple and clean experiment and is easy to implement. It also avoids any legal issues relating to unintended combination effects between different products. On the other hand, it is also very unrealistic and requires simplification of many aspects of the simulation. There is, however, a more important reason not to divorce a pesticide effect from other stressors. That is that the resilience of the population may depend on these stressors. Using the beetle and spider models from ALMaSS this was investigated earlier and found to be a significant factor in the ERA (Topping, Dalkvist et al. 2009). More recently, the phenomenon has been acknowledged by EFSA and resulted in the recommendation of a systems approach to recovery assessment (Committee 2016).

The approach taken here is a compromise. We have kept other stressors in the model in terms of direct or indirect effects of their intended use. For example, an insecticide used as a generic insecticide in the simulation will kill beetles but will have no direct impact on the three other species. It will however, have an indirect impact on skylarks by removing their food. The baseline assumes that no pesticide is used in place of the pesticide being tested. Not withstanding complex emergent effects such as skylarks being influenced by tramlines, then this has no effect on species not affected indirectly or directly by the intended use. For an insecticide this means hare and newt are not affected, for a herbicide beetle and newt are not affected, for a fungicide there is no effect on any of the four species.

6.6 Toxicology in the individual-based model

The method of implementing toxicity in the model used here is to rely on the fact that in the laboratory toxicity is measured over a time period. Typically all that is reported is the effect the length of the study. In the model, toxicity to the individual is based on the assumption that a given toxicology endpoint is measured over a test with a time component. For example we may have an LC_{50} measured over 7 days. The response to the pesticide is built into the model by assuming a threshold concentration above which there is a daily probability of mortality. This probability (^{p}) is calculated from $(1-m) = (1-p)^d$, where m is the proportion assumed to die (e.g. 0.5 for 50% mortality over the test period of 7 days) and d is the number of days over which the test was carried out. If an animal receives a dose of pesticide from the environment then it takes a probability test for mortality against p.

This approach is called the stochastic death model in GUTS TK/TD modelling (Ashauer, O'Connor et al. 2015), and can be contrasted with the individual threshold approach, which sets an individual threshold above which death is certain. The implication of this choice is difficult to determine at the system level, but stochastic death has a larger probability of killing all exposed animals if multiple exposure occurs, whereas at low exposure levels the individual threshold approach leads to higher effects. Both approaches also makes the assumption that an individual that survives exposure does not have any subsequent change in sensitivity (e.g. if it was weakened by the first exposure it might be more sensitive to future exposure). There is no obvious reason to choose one or other approach; however, it is important to make an informed choice in each case.

7. Conclusions and Recommendations

7.1 Conclusions

The main conclusion from this report is that the landscape context is critical in determining the impact of pesticides at population levels. There is variability driven by landscape structure, farming, weather and species ecology and behaviour, with interactions between these. The consequence is that *a priori* selection of realistic worst-case regulatory scenarios is difficult if the aim is to create a single general scenario for all.

To develop landscape-level risk assessment as a reliable tool in regulatory risk assessment it will therefore be necessary to create a set of species models that are agreed and tested. These models should follow the concepts developed the FOCUS for ground and surface water exposure assessment (FOCUS 2001), i.e. have standard interfaces to standard models. However, it is important that the simulations run differ from FOCUS in having a wide range of landscapes and all potential GAPs simulated. Standard landscapes should therefore be developed, which will be facilitated by the methods developed under this project.

The regulatory scenario used for landscape-scale population-level ERA will need to include a specific consideration of the landscape structure, farming, and weather. The implementation of toxicity in the models used also needs to be justified in terms of the stochastic death or individual threshold model of toxic effects on individuals. In addition, it is recommended that the models incorporate multiple life-stage effects in a single simulation rather than assess these separately and combine risk.

Species ecology contributes to the ERA in complex ways and it is not easy to predict from simple metrics such as scale of use, although these metrics correlate well to within species effects.

Other key conclusions from the ERA scenarios are that:

- Impacts of the scenarios were not easily predicted by short-term (i.e. single year) effects, but increased over time;
- Three out of four species demonstrated potential for long-term declines operating over at least two decades;
- Details of ecology can alter the results of the risk assessment, for example the potential for clutch relaying in skylarks interacting with the timing of application;
- Timing of application needs to be considered explicitly;
- Hidden small but long-term effects may be a feature of many systems, and would not be noticed using other methods either because of their short term (field studies) or difficulties with the limit of detection for small effects.

The use of landscape-scale population-level ERA is currently the only way to evaluate long-term effects on populations. However, it is not limited to carrying out individual risk assessments but can also be used to translate laboratory tests to population-level effects, and can be used to help set specific protection goals.

7.2 Recommendations for future work

Currently the approach used here has not been applied to a real risk assessment, therefore many of the issues detected here have not been subject to closer scrutiny, or consideration of how they should be incorporated in the a workable ERA. Therefore, we suggest that definition of the regulatory scenario and all the aspects of context dependency identified in this report should be the focus of future work. Ideally, applying the approach to previously evaluated products to develop methods and scenarios that might be generally applicable to ERA for the different groups requiring assessment (i.e. birds and invertebrates, and probably reptiles and amphibians). A carefully worked and argued example would be extremely useful in future discussions about the implementation of landscape-scale population-level ERA.

The newt model developed under this project has the potential to be used in regulatory risk assessment, not least because EFSA is currently producing a scientific opinion on the need for risk assessment for risks of pesticides to amphibians and reptiles (CJT pers. comm.) However, the model as it stands currently needs improvement before it is ready to be used in this way. To improve the newt model as a tool for regulatory risk assessment we suggest the following actions:

- Development of a sub-model for pond quality prediction from map data
- Development of suitable exposure models for the aquatic newt stages
- Development of a more detailed larval growth model
- Testing of the assumptions of mortality due to weather in the terrestrial stages based on UK conditions for Denmark.
- Further evaluation of model predictions and iterative model improvement.

8. Detailed Methods

8.1 Farming simulation in ALMaSS – an overview

Aspects of the farm management have been described in a number of ALMaSS publications separately (incl. Topping, Ostergaard et al. 2003, Topping, Hansen et al. 2003, Thorbek and Topping 2005, Topping and Olesen 2005, Topping 2011, Parry, Topping et al. 2013), and are described in the program documentation ODdox (Topping, Hoye et al. 2010) format (e.g. Topping 2009), but have not been described as a whole in text form. Since much of the power of ALMaSS for use in environmental risk assessment comes from its ability to handle detailed farm management, a general overview of the processes is provided here.

The farm management in ALMaSS creates a dynamic and emergent pattern of both crop coverage patterns at landscape and field scales but also patterns of farm management activities in time and space. This information is available via the ALMaSS Landscape class interface for any object in the ALMaSS simulations. Thus, the farming module's purpose is to simulate farming realistically at landscape and farm scales and to provide information on vegetation changes and farming activities to the Landscape class.

8.1.1 Class structure (class names in italics)

The overall class handling environmental information in ALMaSS is the *Landscape*. This class contains a map of the landscape represented by homogenous polygons (*LE* (landscape element)) classified into types. Those designated as type field are represented by the *Field* class, and each is linked to the farm that manages it (either based on real information e.g. GLR (see Methods), or specified as the user wishes). All farms are represented by the *Farm* class, and all farms are held in lists managed by the class *FarmManager*, which is instantiated as a class member of *Landscape*. Crops grown on a field are also classes, each is a specific descendent class of the main *Crop* class, e.g. *SpringBarley*. These classes include the implementation of the specific crop husbandry for that crop.

8.1.2 Process overview

This version of the methods assumes the use of the *Farm* class rather than the *OptimisingFarm* class. The *Farm* class cannot really be considered an agent, since they have no goals on which to base decisions but act following predefined rules made variable by the introduction of stochasticity as probabilistic rules. In contrast, *OptimisingFarm* objects are true agents and have goals and more flexible decision and learning behaviour. *Farm* class is the most common usage of ALMaSS since it requires much less information to set up compared to the more complex alternative.

At the highest level of organisation data is used to classify farms (see section Farm Classification 8.2.1), which in turn determines the rotation used by all farms. This information is used, together with the mapped field polygons and their associated farm classified into types, as the basis for determining crop coverage by area at the landscape and farm level. Day-to-day farm management determines the actual cover on fields and in the landscape, crop structure in terms of height and green and total leaf-area index, and farming activities (e.g. ploughing) (Figure 31).

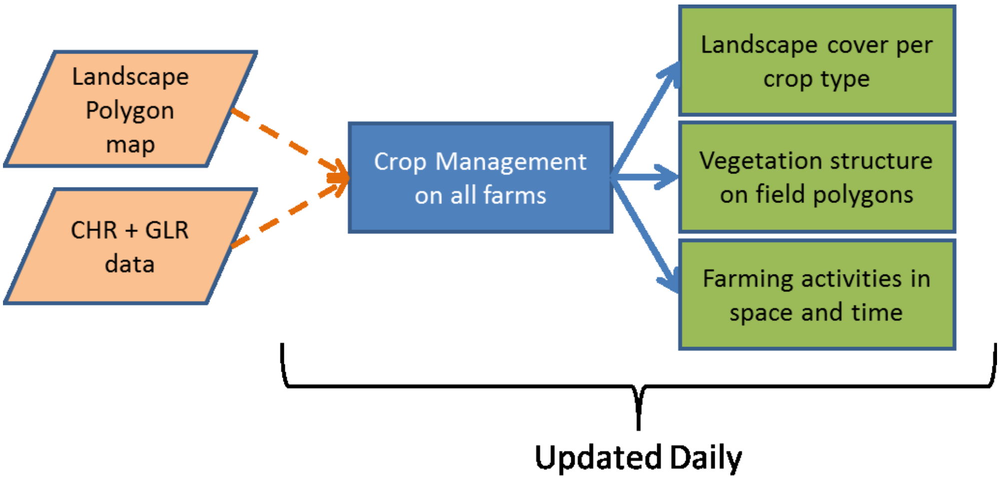


Figure 31: The farm management inputs and outputs at the general level of organisation

On a daily basis the management is carried out at field polygon level and comprises a crop husbandry model fed by system data inputs, linked to a crop growth model and jointly creating outputs at the field polygon level (Figure 32).

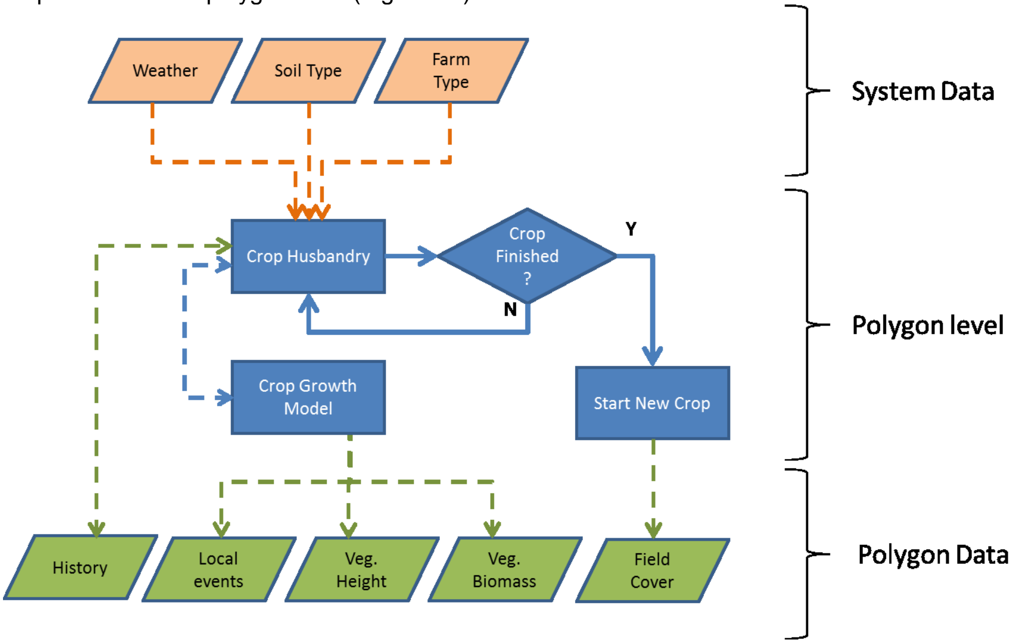


Figure 32: Data flows into and out of farm management at the field polygon level. Dashed arrows are data flows, solid arrows process flow

The basis for the crop growth model is described by Topping and Olesen (2005). This results in a series of growth phases with height, leaf-area total, and leaf area green as a function of summed day degrees from the start of the growth phase (e.g. Table 35). Using these growth phases it is possible to recombine them in different orders to represent all Danish crops e.g. an autumn sown crop will get ‘From Sowing’, ‘From Jan 1st’ ‘From Mar 1st’, ‘From Harvest 1’, whereas a spring crop might be ‘From mar 1st’, ‘From Sowing’. ‘From Harvest 1’. Management e.g. harvest or sowing can cause a change in growth phase and a sudden change in vegetation characteristics (e.g. height after harvest), hence at the start of each growth phases it is possible (but not obligatory) to set the vegetation characteristics to a particular value. For each growth phase the rates of change for the three response variables per day degree (calculated between inflection points on the curve, i.e. rows in Table 41 are stored as vegetation specific growth curves. These specific change rates are applied on a daily basis (Figure 33) and vegetation characteristics updated based on the leaf-area index and height changes (cover and biomass can be calculated from these e.g. using Beer's Law for cover).

Growth Phase	TSum	LAI-Total	LAI-Green	Height
From Jan 1st	0	4.83	0	70
	105	3.85	0	56
From Sowing	99999	0	0	0
From Mar 1st	0	3.85	0	56
	289	4.5	1	65
	695	5	1.25	73
	1017	5.5	0	80
	2619	5.5	0	80
	2787	4.83	0	70
	99999	4.83	0	70
After cutting/harvest 1	0	0.3	0	10
	289	4.5	1	65
	695	5	1.25	73
	1017	5.5	0	80
	2619	5.5	0	80
	2787	4.83	0	70
	99999	4.83	0	70
From cutting/harvest 2	99999	0	0	0
	289	4.5	1	65
	695	5	1.25	73
	1017	5.5	0	80
	2619	5.5	0	80
	2787	4.83	0	70
	99999	4.83	0	70

Table 41: An example of the format of the vegetation growth curve data used in ALMaSS as input to the vegetation daily growth model showing the five growth phases possible.

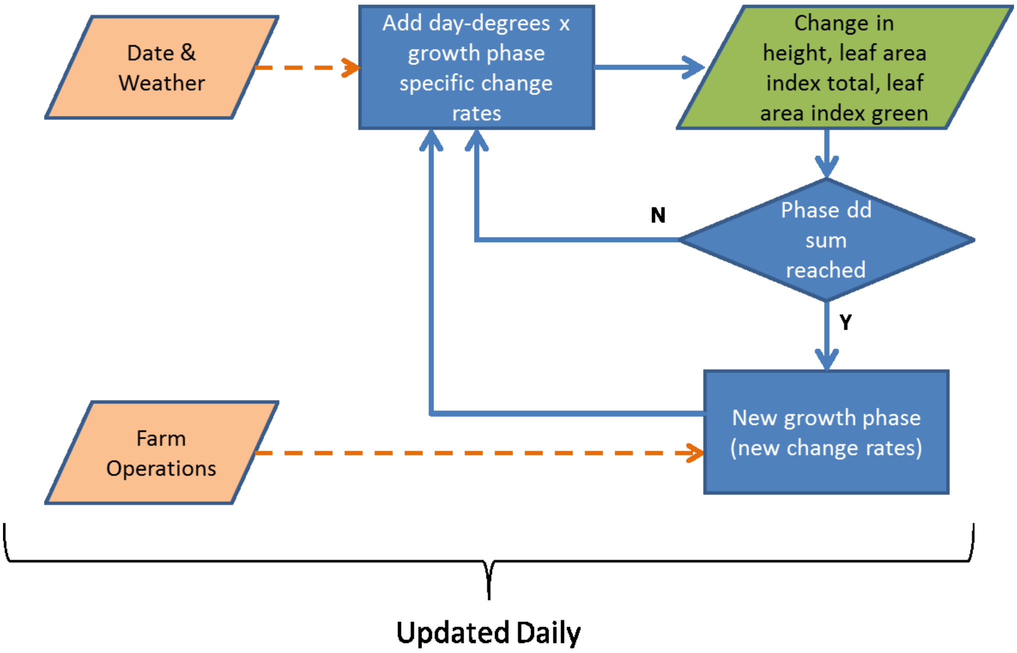


Figure 33: The crop growth model data and process flows leading to changes in vegetation height and leaf area index. Data is represented by dashed lines, process flows by solid lines

Crop husbandry results in recording the farm operation events that occur on each field each day. The sequence of events and their conditions is specified by a unique crop management plan for each crop. Examples of the implementation can be found in the ALMaSS ODdox (e.g. Topping 2009). These plans are long and complicated and hence only a section of a representative plan is shown here to demonstrate the process (Figure 34). Whether an operation is carried out on a particular day is determined by a probability distribution resulting in a distribution of operations in time within the permitted period of action (start to end date inclusive in Figure 34). It is also dependent upon weather or history events e.g. spraying a second herbicide may only happen if the first was sprayed and only under low wind-speed and no precipitation conditions. In this way, real agronomic constraints can be included.

Some constraints for farm operations are programmed into the farm class and are thus general to all attempts to carry out the operation e.g. weather constraints to pesticide sprays; others are specific to the crop husbandry plan and the actual polygon it is being applied to, e.g. history of operations.

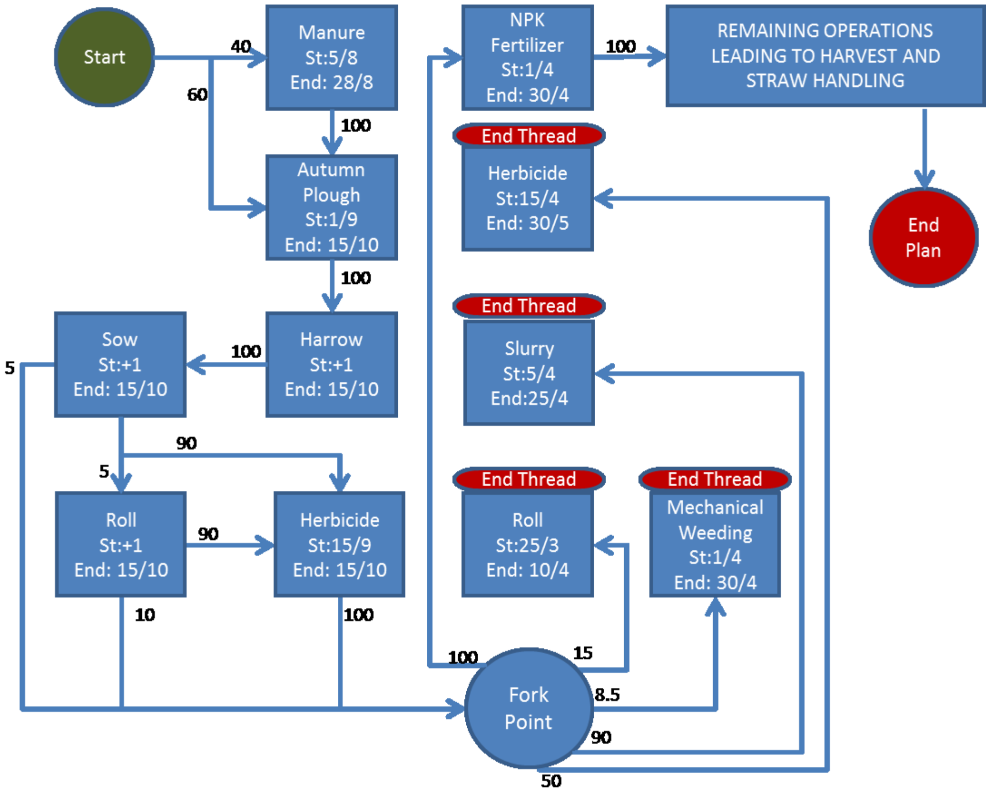


Figure 34: Flow diagram showing events in the husbandry plan for triticale up to end of April. Black numbers are base % making the choice to carry out an operation (blue boxes). Note threads fork from the 'Fork Point' and carry on parallel actions until each thread ends. One thread, in this case the NPK thread carries on to the rest of the husbandry plan (not shown). St = start date End = end date for an operation

8.2 Model landscape creation

The process of generating a complete simulation landscape for ALMaSS is divided into two main tasks:

1. Farm classification (to classify all farms in Denmark into a number of general farm types)
2. Generation of model ALMaSS landscapes

8.2.1 Farm Classification

A program was written in C++ to classify all farms in Denmark into general farm types (http://www.ecosol.dk/MSTProject/Documentation/FarmClassification/index.html). The program classifies all farms based on a combination of the crops they are growing using data obtained from the General Farm Register ("Det Generelle Landbrugsregister" - GLR), and on the animals they have which is data from the Central Livestock Register ("Det Centrale Husdyr Register" - CHR). The GLR is a compilation of the data submitted by the farmers in support of EU subsidy payments. The CHR is a register of all agricultural animals maintained primarily for purposes of disease control.

By combining crop and animal information it was possible to identify major farm types such as pig, arable, or dairy farms. Some less common types are also identifiable e.g. farmers that grow sugar beet on contract. In addition to this information the GLR also indicates whether a farm is organic or not and the overall farm size. This extra information provides the basis for the classification. Rules used to classify the farms were needed to be very general because real farms tend not to fit neatly into pure farm type rules (e.g. many arable farms have grazing because they have some animals e.g. horses or a few animals for their own consumption. The rules used were:

1. Farms with large proportion of vegetables (minimum 0.5) and larger than 2-ha were organic or conventional Vegetable farms, otherwise if small were classified as 'other'.
2. Farms with a proportion of potatoes or sugar beet not less than 20% were Potato or Beet farms respectively.
3. Farms with animal (cows, sheep and pigs) transformed to standard animal units that have fewer than 20 animal units and an area less than 20 ha were designated as Hobby farms (<25 ha is typically part-time or hobby (Levin 2006), so 20 ha will reduce the chance of misclassifying commercial farms).
4. Farms with animal units above 20 and cattle + sheep above 75% they were designated as Cattle farms
5. Farms with animal units above 20 and pigs above 75%, or crop area of grazing pigs above 15% were designated as Pig farms.
6. Farms with animal units above 20, but not pig or cattle farms, were designated as Mixed Stock.
7. Farms with no animals registered but with large areas of grazing were assumed to be either Cattle farms or Mixed Stock farms depending on whether grazing area was above 40% or 20-40% respectively.
8. All remaining farms must have been Arable farms (i.e. large area with few or no animals and little or no grazing).
9. All farms except ‘other’ could be designated as organic or not dependent upon the information on that farm in the GLR giving a total of 17 farm types possible.

For each farm type, the mean proportion of the farm crop area was calculated for each crop. Crops with less than 1% share of the area of a farm type were ignored and the rest used to create a farm rotation for that classification. It was assumed that the rotation could be represented by 100 crops (1 crop for each 1%). The order of crops followed typical agronomic practices and issues such as late harvest leading to impossible sowing conditions were controlled by the built in ALMaSS farm code (see e.g. paper where described in detail). The result is a pattern of changing crops on a field that matches the overall crop distribution pattern for that farm type precisely over 100 seasons. Viewed on a larger scale crop distributions will therefore be overall correct at any point in time, although the actual crop grown on a single field will not replicate reality. This method does not, however, take into account differing soil types between fields, which in reality would restrict some crop distributions.

8.2.2 Generation of ALMaSS simulation landscape mapping

The aim of this task is to generate a land cover raster map with complete coverage; hence all cells must be classified in accordance with their landscape element type. In most cases existing land cover maps are in a coarse spatial resolution (e.g. 100m * 100m; Corine Land cover 2006, EEA 2013) and landscape elements are often broadly categorized (e.g. ESA 2014). Thus for application in individual based modelling, such maps are inadequate. Alternatively manual digitization from areal images can produce maps of sufficient resolution and detail, but this is extremely time-consuming and therefore often not feasible for larger areas (i.e. several square kilometres). However, in many cases highly detailed vector maps are used for e.g. landscape planning and nature conservation purposes and these maps are increasingly becoming publicly available (European Union Open Data Portal – open-data.europa.eu/en/data). Here we make use of a large number of such vector maps and combine them into a single raster map with high spatial resolution as well as with a large number of landscape elements. However, using a large number of different data sources can result in inconsistencies if maps have been made independently and/or if they have been made in different points in time. In most cases these inconsistencies will relate to the spatial alignment of vector layers resulting in either overlaps or gaps between features that are actually adjacent to each other (Figure 35).

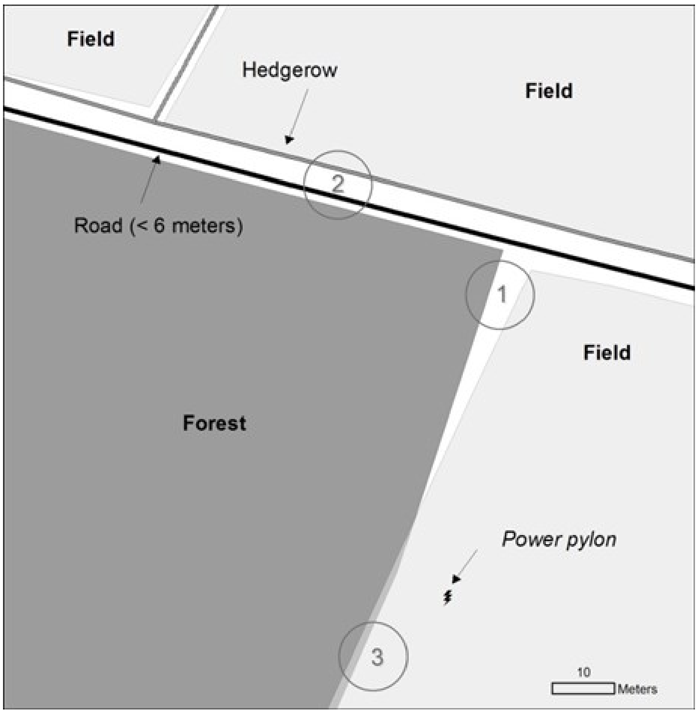


Figure 35: Examples of the three basic vector data types (polygons, lines and points) used as input to the land cover map. Vector data have been digitized for different purposes, in different points in time and they vary in spatial accuracy and detail and in the way the geometry is represented. The figure shows some of the common problems encountered when converting a number of vector themes to a surface covering map. (1) Gaps between polygons. (2) Lack of dimension, i.e. points are per definition dimensionless and lines have length, but not a width. This is obviously a cartographic abstraction and a land cover map need to address the exact extent of a feature (e.g., the width of a road, the actually area covered by a solitary tree etc.). (3) Spatial overlap. Vector layers differ in spatial accuracy which results in gaps or in overlap.

Additionally certain vector types, such as points and lines are dimensionless and therefore decisions about their dimensions needs to be made in order to obtain a meaningful mapping of these in a raster format (Figure 36). When working at high spatial resolution, these issues can be quite substantial and needs to be dealt with in order to obtain a surface covering land cover map. In the remainder of this section we describe methodologies to process a large number of input vector data to produce a surface covering land cover map while taking the abovementioned problems into account.

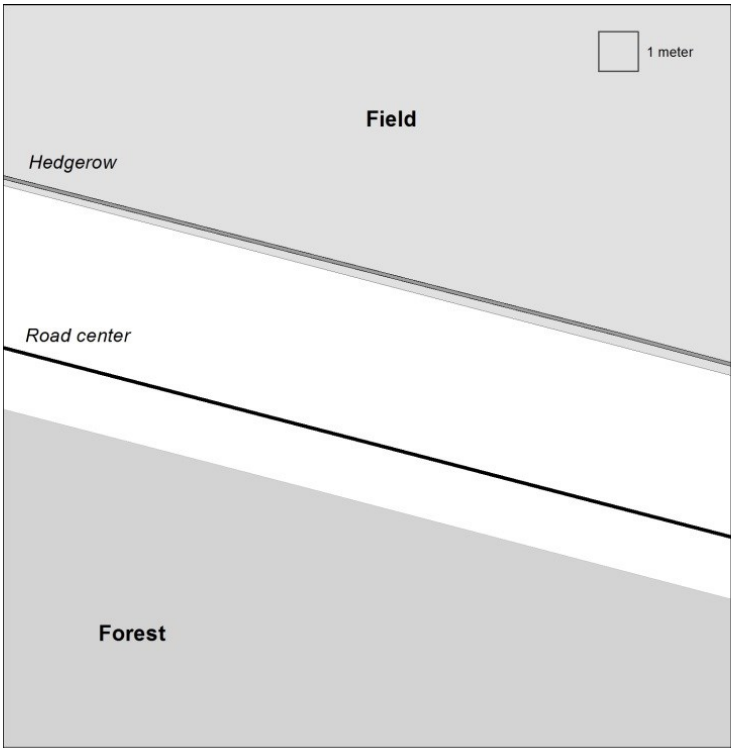


Figure 36: Part of Figure 35 at finer scale showing gaps between polygons and lack of dimension. A forest- and a field-polygon with empty space in between, but two line features (a road and a hedgerow) will supposedly fill this gap. In this particular case, it is known that the road is a medium sized with a width between 3 and 6 meters. There is no further information about the hedgerow. The map reveals some problems: The middle of the road is not centred in the gap between the forest and the field and the centre of the hedgerow is situated inside the field polygon and divides it in two.

The overall process to generate the land cover map follows three steps: 1) Convert the input vector data to raster format, 2) combine individual raster layers into thematic maps (e.g., all road types, paths and railway tracks in a transportation theme), 3) stack these thematic maps in a reasonable order (i.e. roads on top of fields etc.). Below we first describe the input data, and then the three steps involved in generating the final map are described in detail.

8.2.2.1 Input data

We used publicly available vector map layers from the Danish common public geographical administration data (GeoDanmark data, downloaded 2012, http://download.kortforsyningen.dk). The vector layers were used to map 41 different landscape features (see Table 42). The Danish AgriFish Agency (DAFA, under the Ministry of Food, Agriculture and Fisheries) provided vector maps of individual agricultural fields. Lastly, where a pixel was not covered by a field or the GeoDanmark layers (approximately 3-5% of the area, we used the Area Information System (AIS) data, which is a surface covering map with 45 different land cover types for Denmark (Stjernholm et al., 2000). The AIS map is based on data and satellite imagery from the late nineties, but is, nevertheless, the best surface covering map available.

Layer	Description	Theme	Source	Type	Cut-off
landsea	Landmass	All	GeoDK	Polygon	-
slopes_105	Slopes along roads	Road	GeoDK	Line	2.5
roadsideverge_110	Road side verges	Road	GeoDK	Line	1.75
paths_112	Paths	Road	GeoDK	Line	1.51
parks_114	Parking areas	Road	GeoDK	Polygon	3.0
dirtroads_115	Unpaved roads and tracks	Road	GeoDK	Line	2.25
railways_120	Railways	Road	GeoDK	Line	4.5
smallroads_122	Small roads (< 3 meter)	Road	GeoDK	Line	1.75
mediumroads_125	Medium sized roads (3-6 meter)	Road	GeoDK	Line	3.0
largeroads_130	Large roads (> 6 meter)	Road	GeoDK	Line	5.0
pylons_150	Power pylon or transmission tower	Road	GeoDK	Point	1.5
windturbines_155	Wind turbines	Road	GeoDK	Point	1.5
builtuplow_205	Built up areas low	Building	GeoDK	Polygon	-
builtuphigh_210	Built up areas high	Building	GeoDK	Polygon	-
citycenter_215	City center	Building	GeoDK	Polygon	-
industry_220	Industrial areas	Building	GeoDK	Polygon	-
churchyard_225	Cemeteries	Building	GeoDK	Polygon	-
sportsfields_230	Sports areas	Building	GeoDK	Polygon	-
buildings_250	Buildings	Building	GeoDK	Polygon	-
forests_310	Forest	Natural	Top10DK	Polygon	-
shrubs_315	Shrub	Natural	Top10DK	Polygon	-
sand_320	Sand flats	Natural	Top10DK	Polygon	-
heathland_325	Heath land	Natural	Top10DK	Polygon	-
wetland_330	Wetland	Natural	Top10DK	Polygon	-
meadowprotected_355	Protected meadows	Natural	GeoDK	Polygon	3.0
heathlandprotected_360	Protected heath land	Natural	GeoDK	Polygon	3.0
bog_365	Protected bog	WetNature	GeoDK	Polygon	3.0
drygrassland_370	Protected dry grassland	Natural	GeoDK	Polygon	3.0
marshprotected_375	Protected salt marshes	Natural	GeoDK	Polygon	3.0
lakesprotected_380	Protected lakes	WetNature	GeoDK	Polygon	1.0
lakes_440	Lakes	FreshWater	GeoDK	Polygon	-
smallstreams_435	Small streams (< 2.5 meter)	FreshWater	GeoDK	Line	2.01
mediumstreams_436	Medium streams (2.5 - 12 meter)	FreshWater	GeoDK	Line	7.0
largestreams_437	Large streams (> 12 meter)	FreshWater	GeoDK	Line	7.0
lakebuffer_420	Lake buffer	FreshWater	GeoDK	Derived	2.05
fields_1000	Agricultural fields	Fields	DAFA	Polygon	-
dikes_620	Dikes	Cultural	GeoDK	Line	1.2
archeological_625	Archeological sites	Cultural	GeoDK	Point	6.0
recreational_630	Recreational areas	Cultural	GeoDK	Polgon	-
hedgerows_635	Hedgerows	Cultural	GeoDK	Line	2.0
coppice_640	Tree groups	Cultural	GeoDK	Point	8.0
individualtrees_641	Individual trees	Cultural	GeoDK	Point	4.0
gravelpits_650	Gravel pits	Cultural	GeoDK	Polygon	-
ais_1100	AIS landcover map	AIS	AIS	Raster	-

Table 42: Description of the individual layers used in the final map, the theme in which they are grouped, their data sources and the original data type. Layer names follow the convention shortdescription_numericvalue, e.g. the layer slopes_105 is the layer with slopes along roads and 105 is the numeric value indicating presence of the feature.

The Danish AgriFish Agency (under the Ministry of Food, Agriculture and Fisheries) maintains a map of all fields and a database of crops grown in Denmark. The database is updated annually. Farmers are obliged to report for each individual field the crop they intend to grow the following year. The data set used for this study stems from 2013 where more than 45.000 farmers contributed to the database. The data set makes it possible to identify the owner (or manager) of each field and the actual crop grown on it.

The ALMaSS landscape simulator modifies the actual production in each field based on the dominant soil type. The Danish centre for food and agriculture (DCA) at Aarhus University maintains soil classification maps and we used the 1:200.000 map for this study (downloaded from http://dca.au.dk/forskning/den-danske-jordklassificering/). The soil classification map was rasterized and field polygons were overlaid to determine the dominant soil type for each field

8.2.2.2 Making the map

All handling and analysis of spatial data was done using Python 2.7 and the python library arcpy to access ArcGIS features (ESRI 2010). For documentation of each individual arcpy tool used see help.arcgis.com (search for: ‘What is ArcPy?’). The entire process of producing surface covering land cover map has been programmed in a python script that is freely available on Github (https://github.com/flemmingskov/python-landscapegen/tree/PrepForPub). In addition to the data described above an outline of the simulation area is needed before running the python script. The outline needs to be rectangular, in raster format and have the desired spatial resolution (usually 1m by 1m). The outline will be used as a clipping mask to clip any of the data layers that extend beyond the simulation area (Figure 43).

8.2.2.2.1 The Python script

The python script is divided into 5 sections of which the first four makes the land cover map. The fifth is only needed if the map is prepared in order to run ALMaSS simulations. Each main operation is described here with references to figures where relevant.

In the first section (Setup) the libraries needed are imported, paths to input data and outputs are defined, the processing environment is defined and generation of individual themes and layers can be switched on or off. The script assumes a geodatabase to store output (Figure 43, top right).

The second section (Conversion), which constitutes the majority of the script deals with the conversion of the original vector data into raster format. For linear and point features the conversion process involves two steps. First calculate a raster with the Euclidian distance from the features (Figure 37) and second define a raster with the numeric value for the feature at a defined distance from the original vector feature and 0 beyond this distance (Figure 38; see Table 42 for detail about distance used for each of the layers). The numeric value chosen to indicate presence of the feature determines the hierarchy when later combining individual layers into themes. Thus at this stage care must be taken to ensure that numeric values within each theme are sensible. Polygon features are converted directly without adding a buffer, except for a few cases where the original mapping was inaccurate (mostly because the features mapped were difficult to delineate, e.g. the border between swamp and marshland).

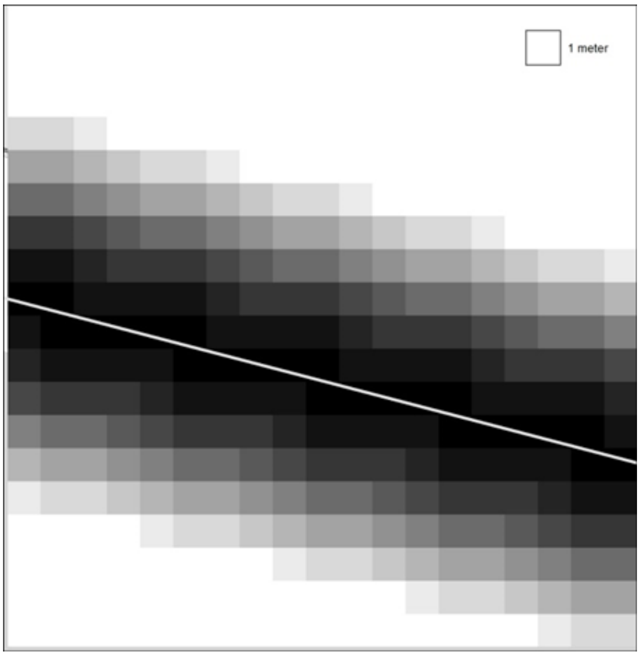


Figure 37: Conversion of a vector feature (here a line) to a raster layer. For each cell in the raster grid the Euclidean distance to the nearest point of the line is calculated (the darker the shading, the closer the cell is to the line). The next step is to choose a cut-off value to select the cells that will be coded as 'road' in the final land cover map.

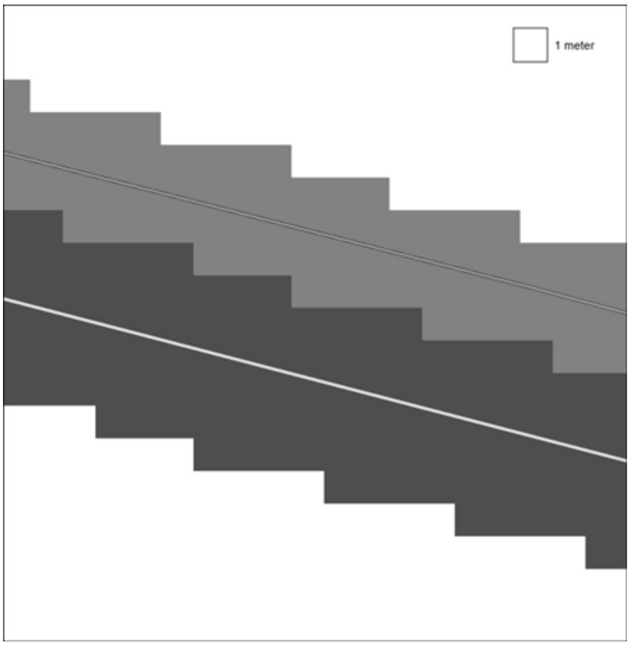


Figure 38: The final representation of 'road' and 'hedgerow' in the land cover map is shown here. The actual choice of a width for a line feature depends on the information available and the purpose of the map. In the present case it is known that the road is medium sized (between 3 and 6 meters); the highest of the two values was chosen and all cells <= 3 meters from the road centre was included as road. No information about the width of the hedgerow is available. Here an average width of 3 meters was chosen.

The third section (Themes) collects the raster layers into thematic maps (e.g., all road types, paths and railway tracks in a transportation theme etc., see Figure 39). In cases where two or more of the layers in a theme overlap, the layer with the highest numeric value is prioritized. For example if a large road (numeric value 130) intersects a small road (numeric value 122) the large road gains predominance and is shown on the final map (Figure 40).

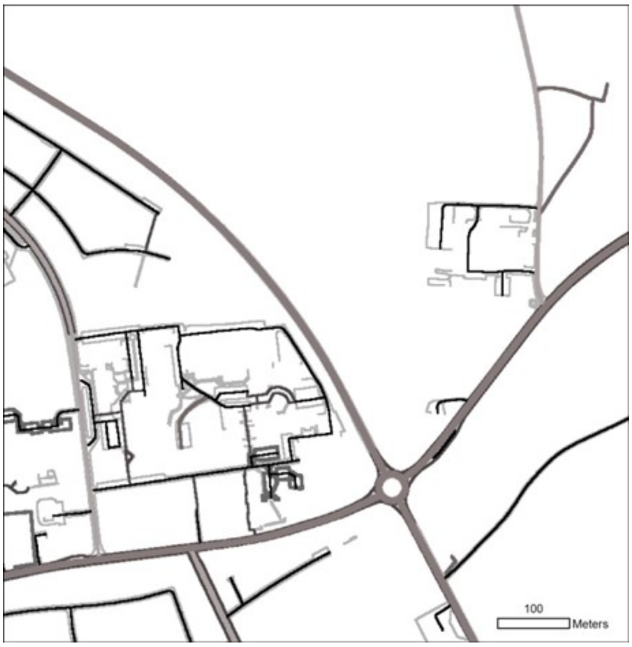


Figure 39: Example of a thematic map. The road network theme is made up of 11 individual raster layers (see Table 42). Each of these raster layers has a numeric code > 0 assigned to cells occupied by the feature in question (and 0 if the feature is absent). The final theme is made by comparing all raster maps cell by cell and choosing the maximum value. Thus, the numeric value rank features and determines which layers will occupy a given cell when more features are present. In the road network theme, for example, small roads (code 122) will precede railway tracks (code 120). The choice of codes is therefore important and depends on the purpose of the final land cover map.

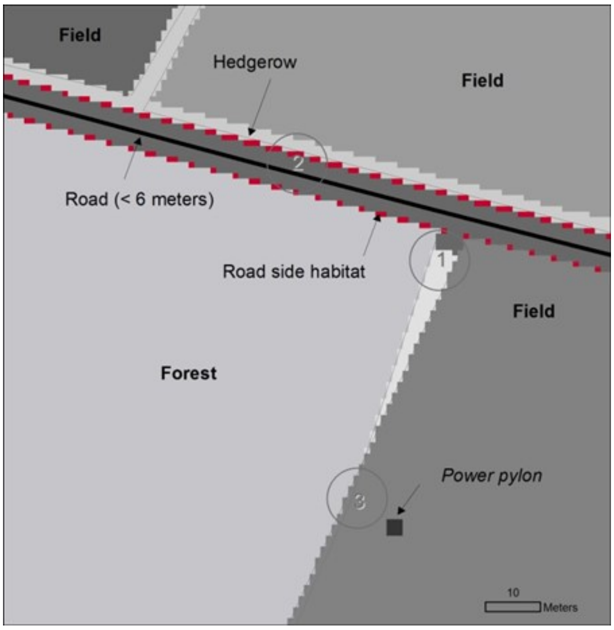


Figure 40: Map assembly and final mosaic map. The individual themes are put together according to a set of rules determining the order of stacking. The order depends on the

purpose of the final land cover map and may be changed accordingly. The numbers refer to the problem areas described in Figure 35. The procedure is shown in Figure 41.

The fourth section (Stack) stacks the thematic maps in sequence such that the final map shows the ecological meaningful layers on top. E.g., the fresh water theme has to be stacked on to the fields to avoid artificial overlaps (Figure 41).

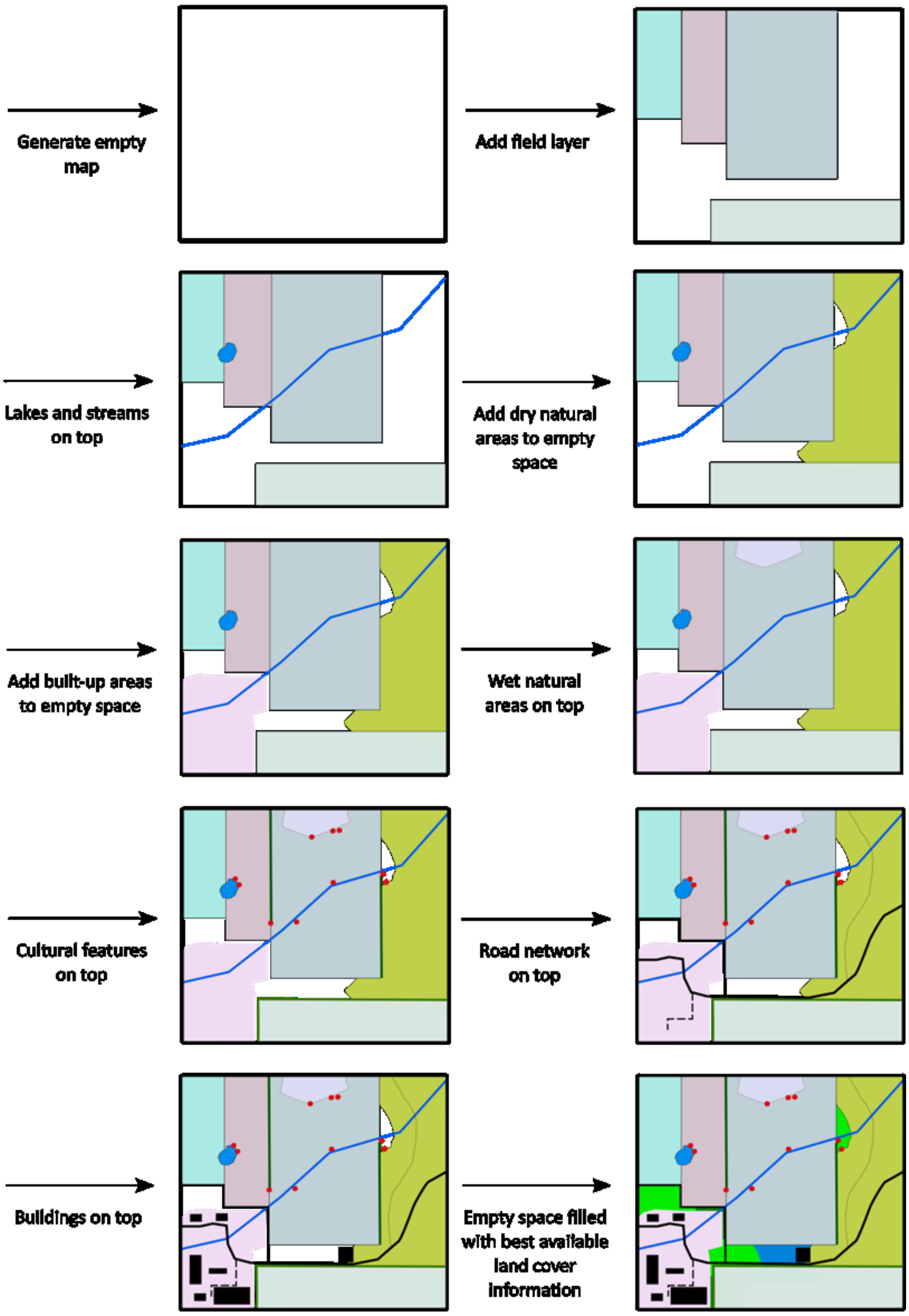


Figure 41: Stacking. Illustration of the stacking procedure used in this study. See text for further detail.

For the present project, the land cover map was assembled using the following rules: (1) The field layer was the bottom layer and (2) the lake and streams (freshwater theme) was put on top of the field layer. (3) Dry natural areas (nature theme) were then added, but only to cells not already occupied by 1 or 2. (4) Similarly, built-up areas (builtup theme) were added to cells not already occupied by 1, 2 or 3. (5) Wet natural areas (wetnature theme), (6) Cultural features (cultural theme), (7) roads (road theme) & (8) sea (raster layer landsea) were then added sequentially onto preceding map. Finally (9) buildings (raster layer buildings_250) were added. After this process there may still number of cells without land cover type, depending of the quality of the input data. There may be several strategies to complete the map: One option is to compare gaps to a recent orthophoto and manually assign a land-use category, but this is a very time-consuming for larger maps. Another option is to fill gaps with a randomly chosen landscape feature (stratified to represent the general landscape structure). For the present study we used an existing, older land cover map of Denmark to fill in gaps.

8.2.2.3 Preparing maps for ALMaSS simulation

8.2.2.3.1 Convert land cover map to ALMaSS landscape

Finalizing the ALMaSS landscape is done in the fifth and last section of the python script (Finalize). The land cover map contains more detail than are used in ALMaSS for most applications. This information has been retained up until now, but will need to be condensed into the landscape element types to be used in ALMaSS. This is a simple reclassification based on a text file with the landscape element codes used in ALMaSS. All features in the raw ALMaSS landscape, both features consisting of single or of multiple raster cells, have a unique value that is common to all cells within the feature. This is achieved by regionalizing (Figure 42 the raster before exporting the map as an ASCII file.

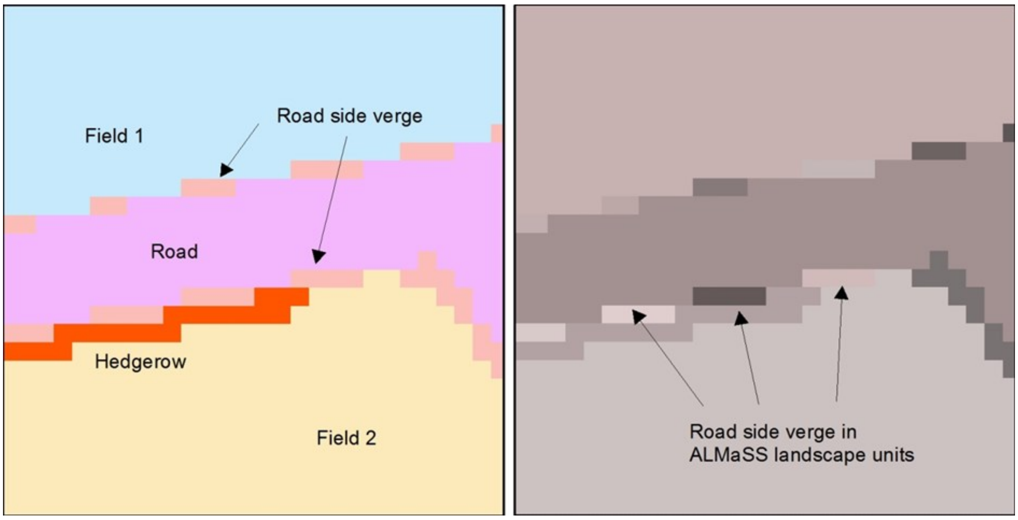


Figure 42: Regionalization. This figure shows how a reclassified landcover maps is transformed into an ALMaSS model landscape. ALMaSS uses landscape units (defined as clusters of pixels of the same landcover type) to store landscape information. Each unit has a unique identifier (as a polygon would) and a landcover code. This is achieve by the regionalize function in ArcGIS. The map on the left shows the classified landcover map (with two fields, a road, road side verge and a hedgerow. The map on the right shows the results of the regionalization. The two fields, the road and the hedgerow are uninterrupted and treated as units. The road side verge, on the other hand, is fragmented and each fragment must be treated as a unique ALMaSS landscape unit.

8.2.2.4 Polygon reference file

Each polygon on the final ALMaSS landscape only contains one value which is the unique ID of the polygon. All additional information about the polygon, such as landscape element type and farm ownership is contained in the polygon reference file. The polygon reference file is a text file containing a unique ID on all polygons in the landscape, the landscape element type of each of the polygons, the number of cells belonging to each polygon, a reference indicating farm ownership and optionally the soil type of each polygon (See Table 43). Upon loading the polygon reference file in ALMaSS, the program adds coordinates for the polygon centroid in the coordinate system used in the simulation. Additionally, columns with openness scores (used when modelling geese) and a unique ID for unsprayed field margins on fields are added.

PolyType	PolyRefNum	Area	FarmRef	UnSprayedMarginRef	SoilType
80	0	2004	-1	-1	-1
110	1	8105	-1	-1	-1
94	2	163599	-1	-1	-1
110	3	3	-1	-1	-1
123	4	278954	-1	-1	-1
110	5	62	-1	-1	-1
20	6	43825	-1	-1	5

Table 43: Table showing the structure of the polygon reference file when first imported to ALMaSS. The first line is indicating the number of lines in the file (here just an example, the actual file will contain many more lines), second line the headers and all following lines the actual values. The value -1 is used to indicate NA.

8.2.2.5 Farm reference file

ALMaSS needs a farm reference file for the each simulated landscape. The file is a text file with two columns, one being the farm reference number (a unique ID for each farm in the landscape) and a column with the farm type of each of the farms (see “Farm classification above”).

8.2.2.6 Making the reference files

To create the polygon reference file, the attribute table from the final land cover map needs to be exported manually from ArcGIS. With this, the attribute table and information about farm ownership (which farm owns which fields) a minimal polygon reference file can be made. If soil type information is available it can be used to improve modelling of growth of vegetation in the simulation. The task is to merge these three pieces of information together. Merging can be done in any standard data base program or programming language and the optimal choice of tool will depend on the format in which farm- and soil type information is available. To prepare the polygon- and farm-reference files we used R and functions in the R package ralmass (Dalby 2015). An example script is provided below (section 7.2.3).

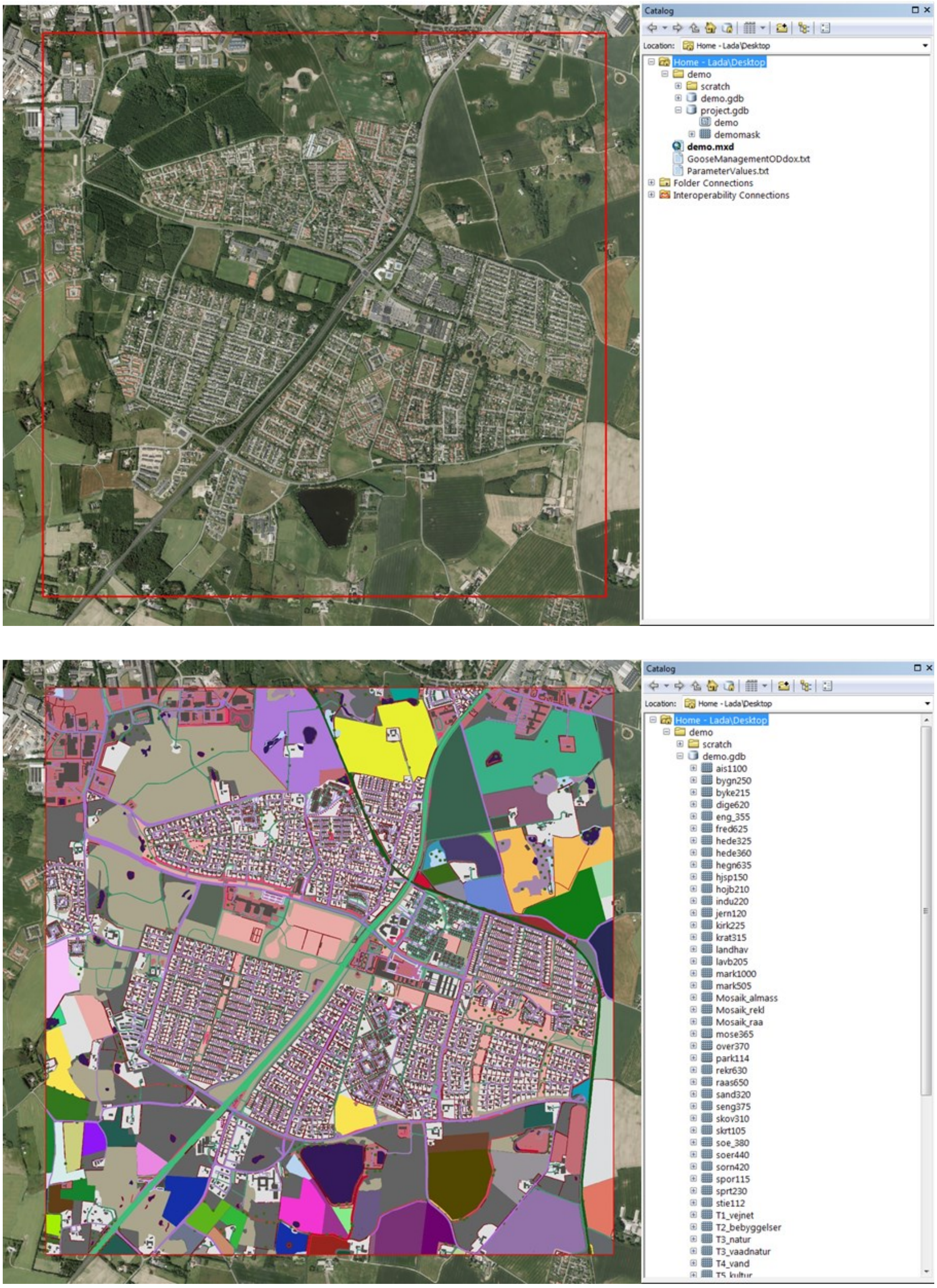


Figure 43: Study area outline and final map. In (a - top) the red outline show the area to make a simulation landscape for. The orthophoto is showing a recent image of the actual landscape. In the catalogue pane on the right hand side of (a) is showing the file structure for the workspace that is needed to run the python script. A Scratch folder to store temporary files, a geodatabase (project.gdb) holding the vector outline (demo, shown in red on in the map view) and the raster version of this outline (here name demomask) and finally a geodatabase to store outputs from the script (here named demo.gdb). In (b - bottom) the final map is shown. In the catalog pane on the right-hand side the demo geodatabase is shown with all its containing layers expanded (some not shown). These are the individual raster layers, the thematic maps and the final maps. They are stored to enable quality control of the individual steps in the process after the final map is made.

8.2.3 R script to generate ALMaSS landscape mapping

Set-up and import the attribute table from ArcGIS.

```
library(devtools)  # Needed to install from github
install_github('LDalby/ralmass')
library(ralmass)
library(data.table)  # We use the data.table package to import the data files
 (some of them are big) as the fread function is very fast
We refer the user to the data.table documentation for detail on the syntax.

# import the attribute table exported from ArcGIS:
PathToFile = 'o:/foo/bar/'
LandscapeName = 'LandscapeX'
FileName = paste(LandscapeName, 'Attr.txt', sep = '')
attr = fread(paste(PathToFile, FileName, sep = ''))
# Use CleanAttrTable in the ralmass package to clean the file:
# see ?CleanAttrTable for documentation
cleanattr = CleanAttrTable(AttrTable = attr, Soiltype = TRUE)
dim(cleanattr)

## [1] 68595     6

setkey(cleanattr, 'PolyType')  # See ?setkey for documentation

# Here we seperate the fields form the rest of the polygons.
# The fields are treated slightly different from the rest.
targetfarms = cleanattr[PolyType >= 10000]  # The fields
targetfarms[,Soiltype:=NULL]  # Remove dummy variable, sinse real soil type d
ata for the fields is available. Will be added further down

cleanattr = cleanattr[PolyType < 10000]  # Everything else
dim(cleanattr)

## [1] 66655     6

str(targetfarms)

## Classes 'data.table' and 'data.frame':   1940 obs. of  5 variables:
##  $ PolyType          : num  13494 13495 13496 13498 13499 ...
##  $ PolyRefNum        : num  6709 5566 212 15020 8580 ...
##  $ Area              : num  1873 58159 11445 8495 8999 ...
##  $ Farmref           : num  -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 ...
##  $ UnsprayedMarginRef: num  -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 ...
##  - attr(*, "sorted")= chr "PolyType"
##  - attr(*, ".internal.selfref")=<externalptr>
```

Next we read in the farm information. In this example the data is stored in a text file where each field is a row in the data set. Each field has a unique ID for the farm to which it belongs.

```
farm = fread('o:/foo/bar/FarmInfo2013.txt')

farminfo = farm[, c('AlmassCode', 'markpolyID', 'BedriftID', 'BedriftPlusID',
'AfgKode'), with = FALSE]  # Extract only the columns we need for now
farminfo[,markpolyID:= gsub(pattern = ',', replacement = '',
          x = farminfo$markpolyID, fixed = FALSE)]  # Fix seperator issue

setkey(farminfo, 'markpolyID')
```

In this particular case we do have soil type information for the fields, so we load that.

```
soil = fread('o:/foo/bar/Soil_type.txt')

setnames(soil, old = 'MAJORITY', new = 'Soiltype')
setkey(soil, 'markpolyID')
```

With all three pieces of information we merge the datasets using the unique field polygon ID as key.

```
SnF = merge(farminfo, soil, all.x = TRUE)
temp = merge(x = targetfarms, y = SnF, all.x = TRUE)
# Rearrange and remove obsolete columns
temp[,PolyType:=AlmassCode]
temp[,AlmassCode:=NULL]
temp[,Farmref:=BedriftID]
temp[,BedriftID:=NULL]

result = rbind(cleanattr, temp)  # This is essentially putting the fields and
everything else back together.

# Check that the dimensions match the original input:

dim(attr)

## [1] 68595     3

dim(result)

## [1] 68595     6  # Okay (we added the extra columns)

setkey(result, 'PolyRefNum')
FileName = paste(LandscapeName, 'PolyRef.txt', sep = '')
WritePolyref(Table = result, PathToFile = paste(PathToFile, FileName, sep = '
'))  # The
 function WritePolyref ensures that the resulting complies with the format re
quired by
 ALMaSS. see ?WritePolyref for documentation.

#-----------------------------------------------#
# Make a farmref file to go with the landscape
#-----------------------------------------------#

# The file The2013Farmref.txt contains all farms in Denmark with their unique
farm reference number and their farm type. See the section ”Farm classificati
on” in Topping, Dalby et al. (2016) for details on the classification. We mak
e a simple subset of this file to only the farms actually situated in the lan
dscape in question.
farm = fread('foo/bar/The2013Farmref.txt')
setnames(farm, c('Farmref', 'FarmType'))
landscapefarms = farm[Farmref %in% unique(result[,Farmref]),]
FileName = paste(LandscapeName, 'Farmref.txt', sep = '')
WritePolyref(Table = landscapefarms, PathToFile = paste(PathToFile, FileName,
sep = ''), Headers = FALSE, Type = 'Farm')
```

8.3 Model landscapes used for simulation

Ten model landscapes were selected to be used in the simulation runs Figure 40. These were designed to represent the range of agriculture present in Denmark, and also to represent a range of landscapes from very intensive agriculture to extensive agriculture with many non-agricultural areas (Figure 44-Figure 54).

Figure 44: The location of each of the 10 landscapes in Denmark.

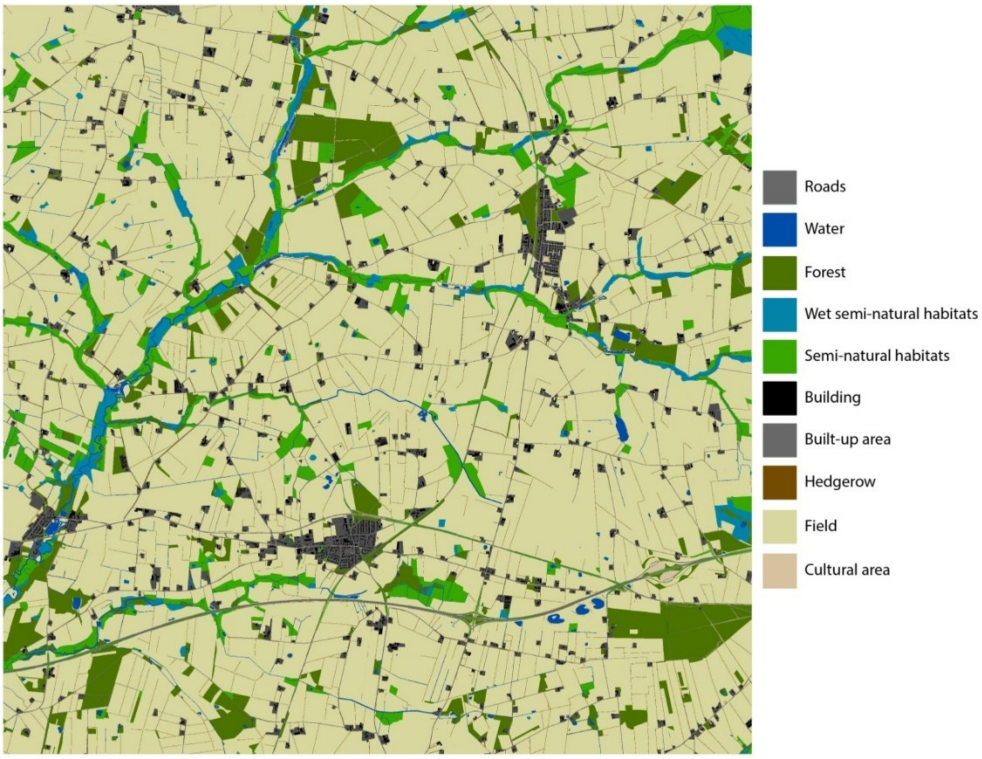


Figure 45: Esbjerg (Es)

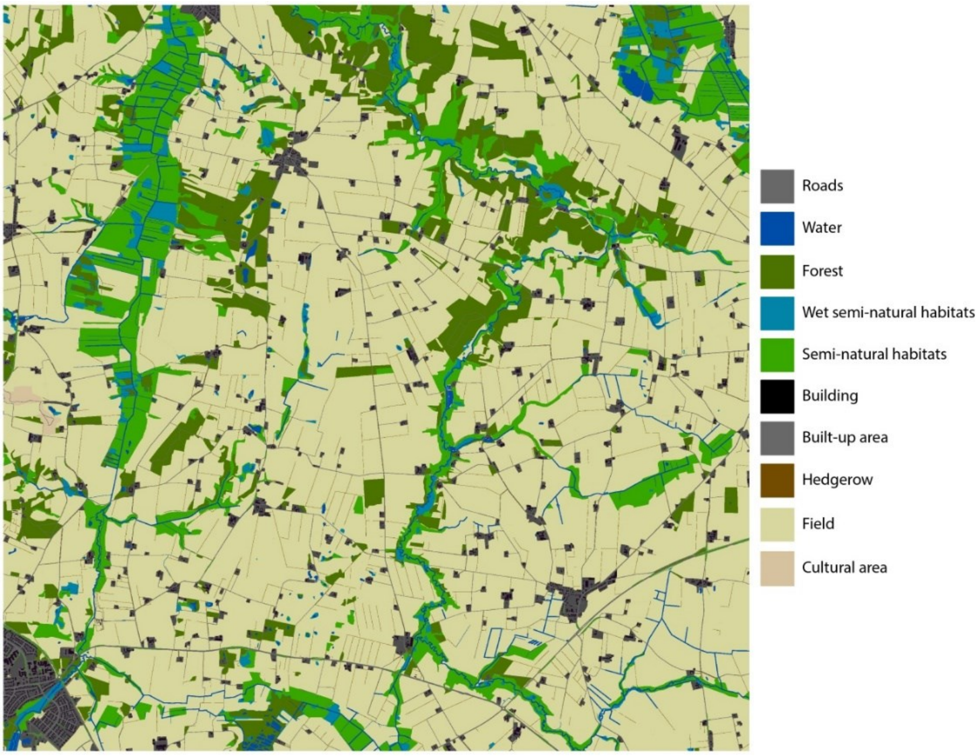


Figure 46: Himmerland (Hi)

Figure 47: Karup (Ka)

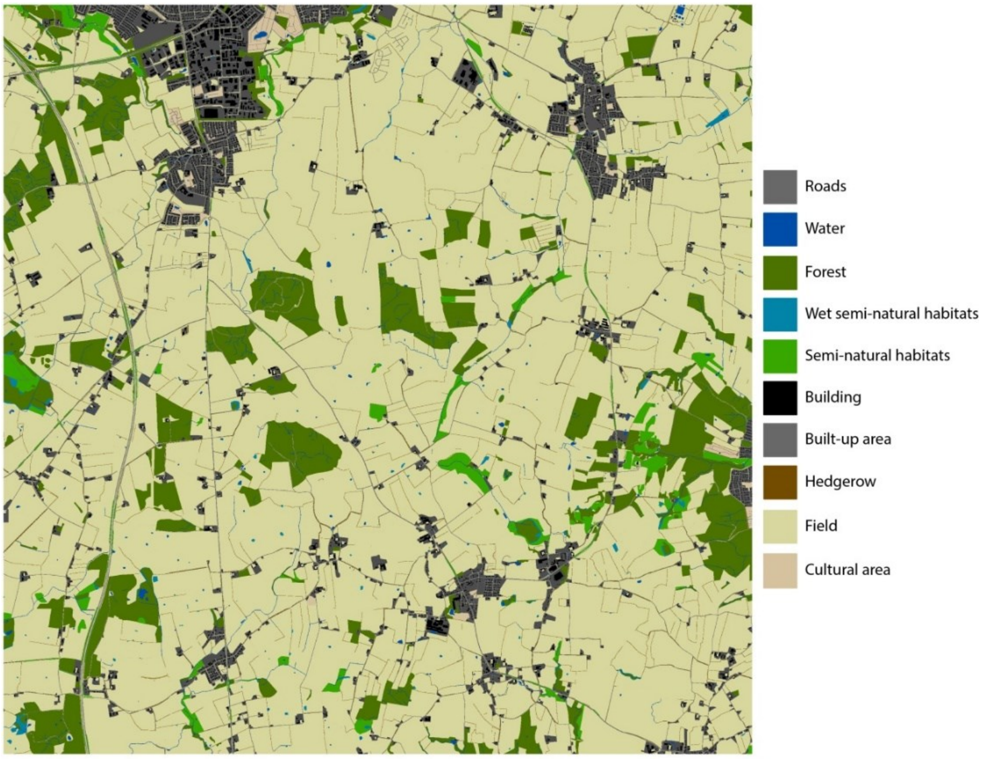


Figure 48: Kolding (Ko)

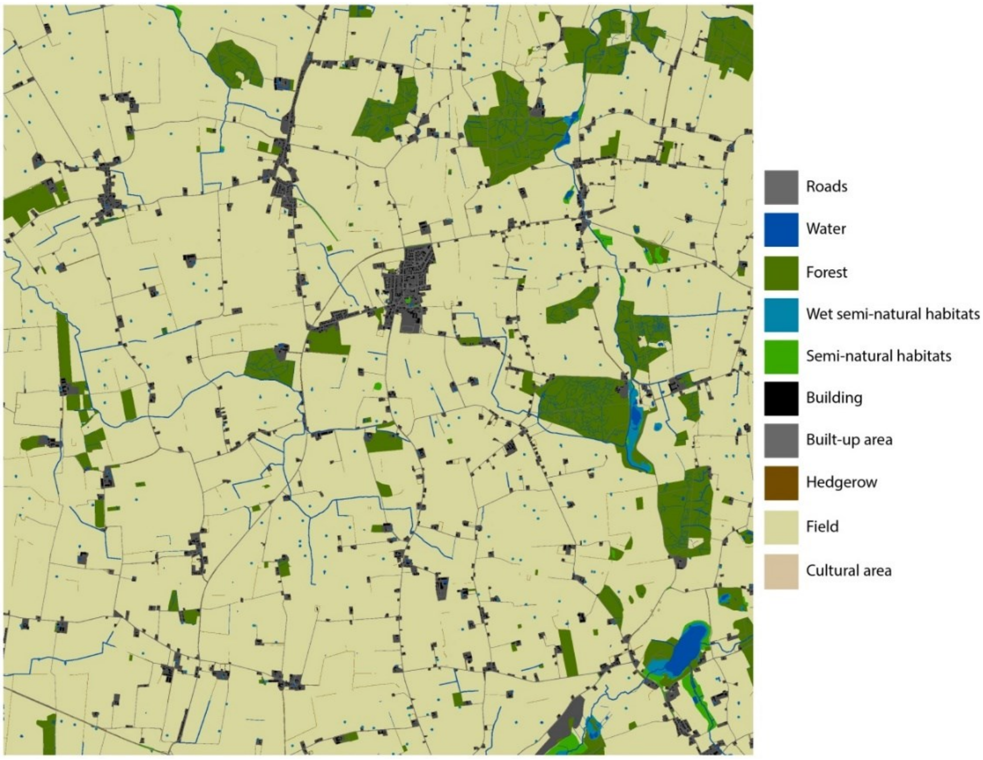


Figure 49: Lolland (Lo)

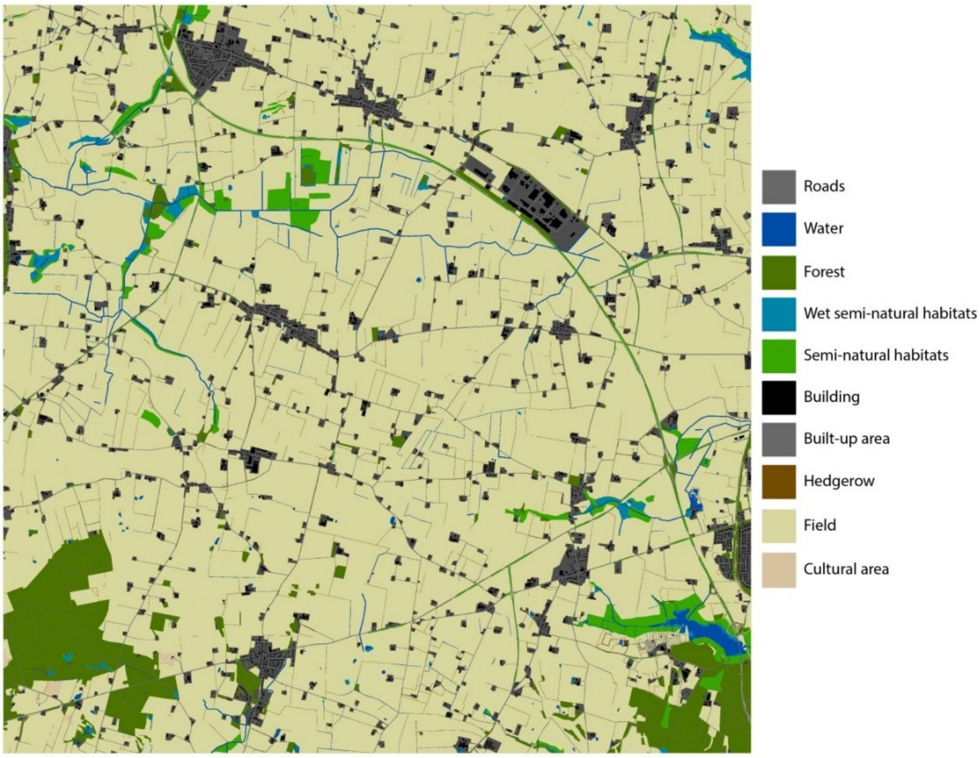


Figure 50: Mors (Mo)

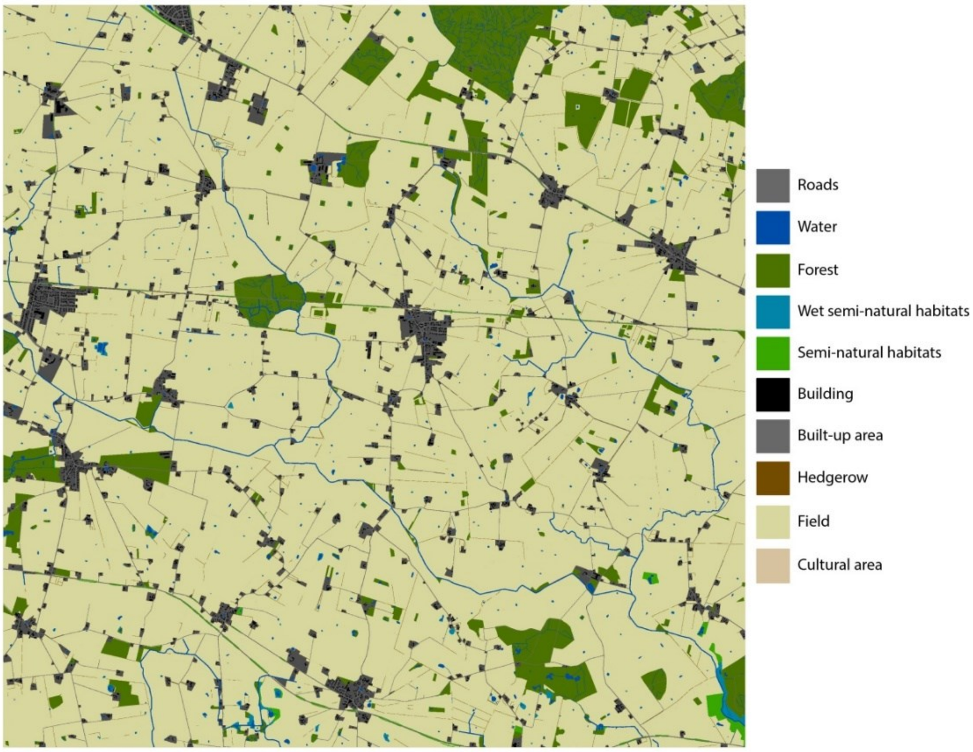


Figure 51; Næstved (Na)

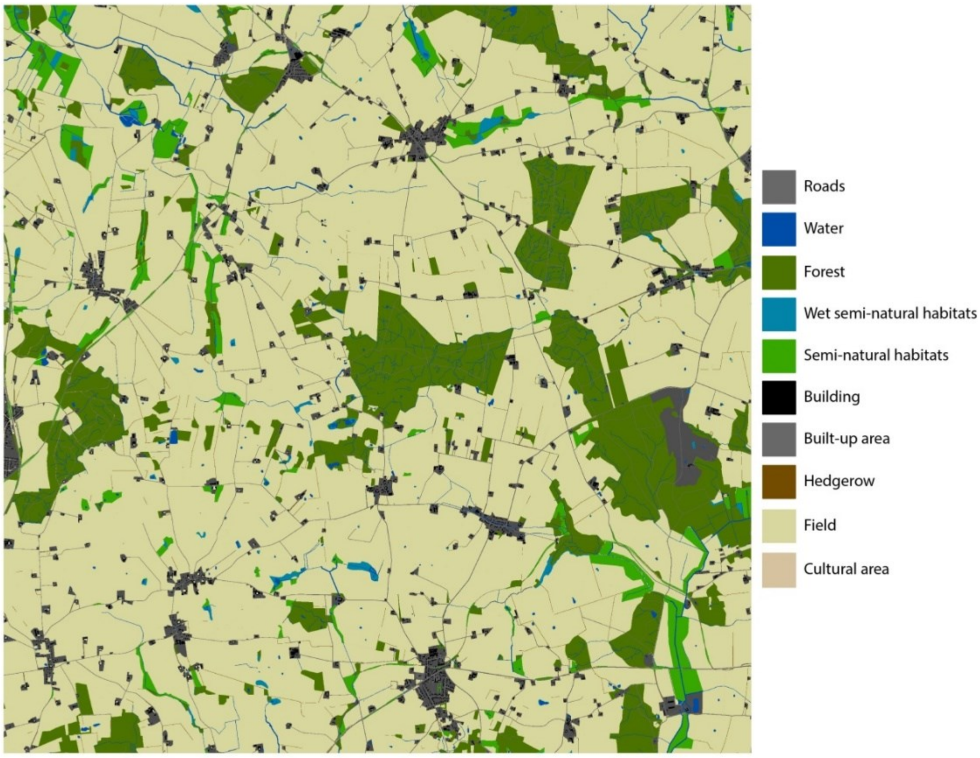


Figure 52: Odder (Od)

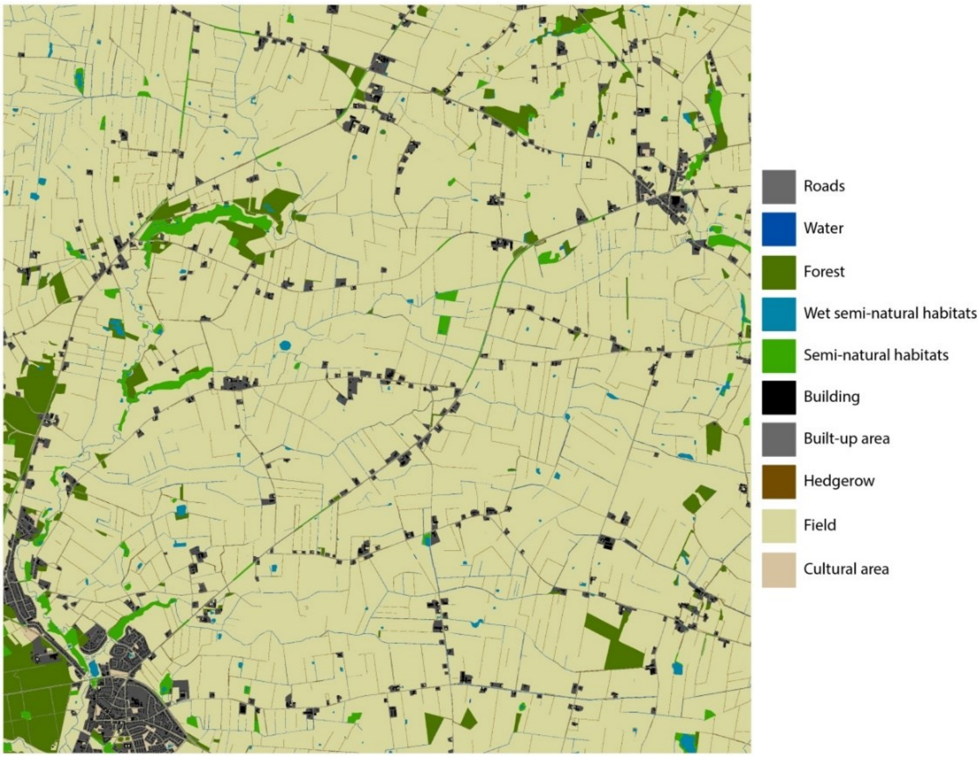


Figure 53: Toftlund (To)

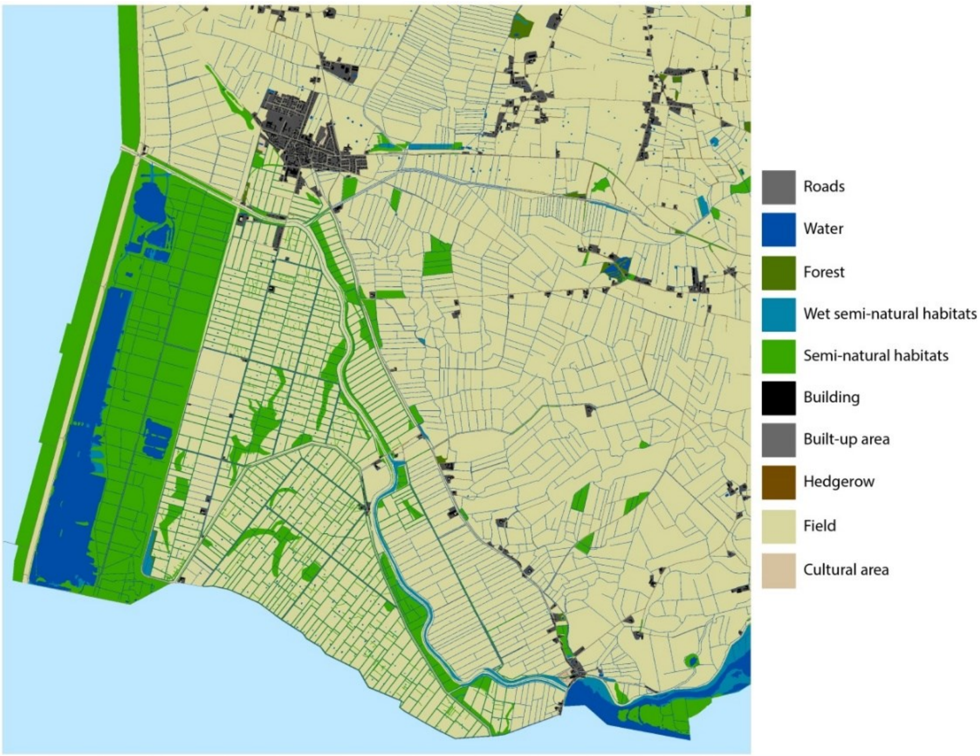


Figure 54: Tønder (Tn)

The proportion of each ALMaSS crop type assumed to be grown by each farm type was based on the national farm classification. The resulting distribution of crops by area to farm types, total areas of crop and pasture and crops by area to landscape are presented in Tables 44-49.

Farm Type/Crop	Conv. Pig	Conv. Cattle	Conv. Arable	Conv. Hobby	Conv. Mixed Stock	Conv. Potato	Conv. Beet	Conv. Veg	Other Farm Types
Maize Silage	0.01	0.13	0	0.01	0.02	0.03	0.01	0	0.01
Oats	0.02	0.01	0.03	0.02	0.02	0.01	0	0.01	0.01
Permanent Grass	0.04	0.17	0.02	0.24	0.07	0.02	0.01	0.02	0.36
Potatoes	0	0	0	0	0	0.18	0	0.03	0.02
Rotational Grass	0.03	0.29	0.02	0.25	0.1	0.04	0.02	0.04	0.31
Seed Grass	0.03	0.01	0.01	0	0.08	0.02	0.02	0	0
Spring Barley	0.28	0.19	0.34	0.22	0.3	0.32	0.35	0.07	0.08
Spring Barley Silage	0	0.05	0	0.01	0.01	0	0	0	0.01
Spring Wheat	0.01	0	0.01	0	0.01	0.01	0	0	0
Sugar Beet	0	0	0.01	0	0.01	0	0.27	0	0
Vegetable crops	0	0	0	0	0	0	0	0.22	0
Winter Barley	0.08	0.02	0.07	0.02	0.04	0.01	0.01	0	0
Winter Rape	0.1	0.02	0.1	0.02	0.07	0.01	0.02	0	0.01
Winter Rye	0.04	0.01	0.04	0.02	0.04	0.03	0	0	0
Winter Wheat	0.32	0.07	0.32	0.09	0.19	0.07	0.28	0.02	0.02
Others	0.02	0.03	0.02	0.08	0.04	0.24	0.01	0.58	0.16

Table 44: Conventional farm types generated from the farm subsidy and animal health databases for 2014, and the proportion of each crop or group of crops assumed to be grown by that farm type.

Farm Type/Crop	Org. Pig	Org. Cattle	Org. Arable	Org. Hobby	Org. Mixed Stock	Org. Potato	Org. Beet	Org. Veg
Maize Silage	0.01	0.02	0	0	0.01	0.02	0	0
Oats	0.07	0.07	0.28	0.07	0.15	0.08	0	0.01
Permanent Grass	0.06	0.17	0.02	0.2	0.07	0.03	0.02	0.02
Potatoes	0	0	0	0	0	0.28	0	0.03
Rotational Grass	0.14	0.4	0.05	0.42	0.17	0.11	0	0.04
Seed Grass	0.03	0.02	0.01	0.01	0.04	0.01	0	0
Spring Barley	0.28	0.09	0.27	0.1	0.22	0.1	0.44	0.01
Spring Barley Silage	0.1	0.09	0.01	0.04	0.03	0.03	0	0.01
Spring Wheat	0.03	0.02	0.06	0.03	0.07	0.04	0	0.01
Sugar Beet	0	0	0	0	0	0	0.24	0
Vegetable crops	0	0	0	0	0	0	0	0.04
Winter Barley	0.02	0	0.01	0	0	0	0	0
Winter Rape	0	0	0.01	0	0	0	0	0
Winter Rye	0.06	0.04	0.12	0.04	0.1	0.08	0	0
Winter Wheat	0.03	0.02	0.06	0.01	0.04	0.03	0.3	0
Others	0.19	0.05	0.11	0.08	0.09	0.18	0.00	0.83

Table 45: Organic farm types generated from the farm subsidy and animal health databases for 2014, and the proportion of each crop or group of crops assumed to be grown by that farm type.

Landscape	Ha	%
Es	7584	76%
Hi	7393	74%
Ka	6618	66%
Ko	6814	68%
Lo	8125	81%
Mo	7747	77%
Na	7885	79%
Od	7243	72%
To	8296	83%
Tn	6227	62%

Table 46: The total area and percentage of area comprising fields and pasture per landscape used.

Crop	Es	Hi	Ka	Ko	Lo	Mo	Na	Od	To	Tn
Maize Silage	0.099	0.1	0.028	0.02	0.006	0.032	0.015	0.025	0.066	0.094
Oats	0.016	0.019	0.014	0.026	0.004	0.023	0.024	0.025	0.038	0.034
Permanent Grass	0.144	0.019	0.139	0.037	0.194	0.012	0.082	0.009	0.03	0.016
Potatoes	0.002	0.001	0.139	0.003	0.001	0.001	0.002	0.001	0.01	0.002
Rotational Grass	0.243	0.256	0.063	0.078	0.018	0.099	0.068	0.085	0.266	0.291
Seed Grass	0.015	0.021	0.014	0.015	0.017	0.023	0.023	0.02	0.018	0.016
Spring Barley Silage*	0.038	0.04	0.009	0.009	0.001	0.012	0.007	0.01	0.05	0.051
Spring Barley*	0.214	0.25	0.271	0.304	0.281	0.288	0.301	0.301	0.225	0.228
Spring Wheat	0.005	0.006	0.005	0.01	0.002	0.009	0.009	0.009	0.013	0.01
Sugar Beet	0.002	0.003	0.001	0.005	0.194	0.004	0.008	0.005	0.002	0.002
Vegetable crops	0.002	0.002	0.002	0.003	0.003	0.003	0.004	0.003	0.002	0.002
Winter Barley	0.027	0.035	0.014	0.063	0.014	0.069	0.056	0.069	0.03	0.029
Winter Rape*	0.031	0.042	0.016	0.086	0.021	0.084	0.082	0.088	0.036	0.034
Winter Rye	0.02	0.025	0.026	0.037	0.005	0.037	0.036	0.038	0.032	0.028
Winter Wheat*	0.11	0.145	0.069	0.276	0.23	0.275	0.26	0.284	0.128	0.123
Others	0.031	0.036	0.191	0.029	0.009	0.030	0.027	0.028	0.053	0.038
Total	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000

Table 47: The proportion of each ALMaSS crop by area in each of the ten landscapes used for the study. * are crop types used for pesticide application in the WSO scenarios (see below).

8.4 Simulated Species and Models

8.4.1 European brown hare

8.4.1.1 Species description

The brown hare has been in widespread decline throughout Europe since the 1960s (Flux and Angermann 1990, Homolka and Zima 1999, Edwards, Fletcher et al. 2000, Smith, Vaughan Jennings et al. 2005, Smith and Johnston 2008). Although present across a wide geographic range, the brown hare is listed under Appendix III of the Bern Convention in Europe (Smith and Johnston 2008), and several countries have placed the species on their Red List as “near threatened” or “threatened” (Reichlin, Klansek et al. 2006). Located in the European cultural landscape, the hare is a typical example of many other open farmland species in Europe that are affected by agricultural intensification and its side effects (Donald, Green et al. 2001, Stoate, Boatman et al. 2001). The typical activity range of a hare is large, more than 20 hectares depending on the landscape type (Schai-Braun and Hackländer 2014), hence the brown hare is an excellent species to examine agricultural changes on larger landscape scales.

Numerous studies show that monocausal explanations of hare population dynamics are not possible (Marboutin, Bray et al. 2003, Schmidt, Asferg et al. 2004, Smith, Jennings et al. 2005). Thus, assessments and evaluations of hare population trends are difficult to perform due to the interactions that occur between multiple stressors and the spatial and temporal variability in field data (Smith, Vaughan Jennings et al. 2005, Topping, Høye et al. 2010). Furthermore there is still a lack of long-term and large-scale population data, despite extensive observation efforts in recent decades (Strauss, Grauer et al. 2008). To understand the ecological significance of agricultural pesticides on hare populations, habitat use must be examined precisely in space and time (Rühe and Hohmann 2004, Smith, Jennings et al. 2004, Strauss, Grauer et al. 2008). This can be achieved using the ALMaSS European Brown Hare model (Topping, Hoye et al. 2010).

8.4.1.2 Model overview

The model simulates the growth, movement, reproduction, and mortality of individual hares using a daily time-step for most activities, but a 1 minute time-step for foraging. Full details of the hare model are described by Topping (2009) in ODDox format, but a short description is presented here to aid readability.

The hare model simulates five life-stages: *infants* up to 11 days during which they are totally dependent on the lactating doe; *young* 12-35 days old after which they are fully weaned, *juveniles* 35-365 days old, adult *males* and *females*. In the model hares are quite mobile and able to find suitable forage over a wide area when not encumbered with young. If feeding conditions are good hares will generally drift over large areas, in poor feeding conditions the hares will optimally forage, and thus may become restricted to localised ranges for periods within a season. Breeding starts in spring if body condition allows for the production of foetal mass. After birth the female must increase her energy intake in order to provide enough energy for lactation. Energy comes from foraging from green shoot material and the amount of energy obtained depends on the age of the shoot and the overall structure of the vegetation. Dense vegetation may therefore have a high food value in terms of biomass but a poor digestibility and high impedance. A female that cannot support lactation because her combined energy intake and reserves fall too low will abandon her young. Reproduction will not be attempted again until energy reserves are replenished. Growth of model hares is also dependent on energy balance and hares which do not achieve 45% of their potential weight at any age will die. Adults rarely die of energy shortage and are assumed to be able to "carry" a negative energy balance. They thus will remain in the population contributing to social stress, which is ultimately the primary density-dependent regulation factor in the hare model and reducing population growth. Hunting occurs in autumn but other non-energetic related losses are based on life-stage specific constant daily probabilities (e.g. predation of young), or on events driven by human management activities (e.g. harvest mortality).

8.4.1.2.1 Pesticide effects

The model hares must forage realistically from the landscape and are exposed to pesticide residues in the process. Foraging is done by selecting a 10 x 10 m area at the current hare location and foraging from this; this is assumed to take 100 minutes. The hare then selects the 10 x 10 m area adjacent to this first forage area that provides the best energetic return based on forage quality and impedance. Time to walk between squares and sample them is also included. This process continues until either the period of time allocated for foraging is used up, or the hares cannot eat any more.

Hares have an energetic maximum daily intake limit of 5500 kJ day^{-1} (Valencak, Tataruch et al. 2009), but stomach contents of wild hares contain 11 kJ g^{-1} (Hacklander, Tataruch et al. 2002), which suggests a daily throughput maximum of 500 g. The maximum rate observed for hares by Andersen (1947) was 1.7g per minute would result in only 294 minutes foraging. This is much less than assumed in ALMaSS where 67% of the daily activity will be foraging or movement associated with foraging. To compensate we assume that intake rate in grams per minute is 500/(1440 min x 0.67) = 0.518 g per foraging minute. If the hare uses less time than this due to other activities, then the pesticide intake rate will decrease proportionally with the forage intake. The ingestion rate of pesticide (mg minute^{-1}) is therefore the environmental concentration (mg/g) multiplied by 0.518 for each minute spent foraging in each location.

The model includes internal and external toxicokinetics (TK) in terms of the varying rates of ingestion of the pesticide, and the process of elimination within the hare. The internal TK are represented by a single compartment model assuming a percentage elimination rate per day. External TK is determined by the feeding behaviour of the hare and ultimately by the time spent feeding from contaminated areas, and the concentration of pesticide on vegetation.

We based the scenarios on a generalised pesticide. Application rates were assumed to be 10 g a.i. per hectare. This gives a residue of 0.4 mg/kg vegetation immediately after spraying based on a mean residual unit dose (RUD) for cereals and leafy forage crops (Fletcher, Nellessen et al. 1994). This will result in a daily dose of 0.2 mg a.i. per day (0.5 kg/d x 0.4 mg/kg = 0.2 mg/d), if an adult hare eats its full 500 g from a contaminated area immediately after spraying. We can calculate that the initial rate of intake per foraging minute after spraying would be 0.000518kg * 0.4 mg/kg = 0.000207 mg/min.

For the chronic effect of the pesticide, the impact of exposure above a threshold body-burden was modelled as a uniformly distributed chance of litter size reduction of 0-100% for female hares exposed during gestation of that litter. Initial scoping runs indicated that a trigger threshold of 0.0001 mg a.i./kg bw with an internal degradation rate of 5% per day gave noticeable population level impacts. This was designated as the 1X toxicity scenario.

8.4.2 Skylark

8.4.2.1 Species description

The skylark *Alauda arvensis* is one of the farmland bird species for which population declines have been most severe throughout most of Western Europe (Fuller, Gregory et al. 1996, Siriwardena, Baillie et al. 1998, Chamberlain and Crick 1999). Being one of the few farmland birds to nest and feed almost exclusively in open fields and field margins, the skylark is one of the farmland bird species most likely to be vulnerable to changes in crop management. A number of studies have attempted to identify the causes of decline in the skylark. They have demonstrated a negative influence on skylark reproduction of tall, dense and fast-growing autumn-sown cereals, a simplification of crop rotation leading to a decrease in crop diversity and structure (Odderskaer, Prang et al. 1997, Wilson, Evans et al. 1997, Wakeham-Dawson, Szoszkiewicz et al. 1998, Chamberlain, Wilson et al. 1999), silage cutting and trampling on grass fields (Wakeham-Dawson, Szoszkiewicz et al. 1998), and loss in winter stubble fields (Donald, Buckingham et al. 2001).

The skylark is a ground nesting bird arriving in Denmark early in the year, with males arriving generally before females and setting up defended territories. Territories will be in open land, where vegetation allows the construction and access to nests made from grass on the ground. Very dense homogenous vegetation is unsuitable, as are sites near to tall structures. Eggs are laid in clutches of 3-6 eggs from March to June. Incubation is by the female and takes 10-17 days. After egg hatch, the chicks are fed a diet of arthropods by both parents. Chicks can thermoregulate after 5 days and nest leaving usually takes place after 8 days (but can be up to 11). Following nest leaving, the juvenile bird is fed primarily by the male typically until 31 days old, after which it is independent.

8.4.2.2 Model Overview

The ALMaSS skylark model has been used in pesticide risk assessment for some time (Topping and Odderskaer 2004, Topping 2005, Topping, Sibly et al. 2005). The most recent model version and its testing is described in Topping, Odderskaer et al. (2013), and full documentation for the skylark model can be found in Topping (2011). As with the hare model an overview is presented here to aid readability.

The individual model skylarks are categorised as being members of five life-stages, clutch, nestlings, pre-fledglings, males, and females. The main drivers of the skylark model are the topography and habitat quality of the landscape elements being modelled, farming activities (crop choice, physical disturbance), crop growth, and weather. Available insect food biomass is determined by vegetation structure in each landscape element and type (i.e. locally for each patch), see (Topping 2012), and by its availability in terms of physical accessibility to the birds during foraging. Insect biomass resources are updated daily in the model and are affected by vegetation growth processes and also by human management (e.g. insecticides or herbicides). During the breeding period, defined here as incubation and care of young up to 30 days old, the model considers the energetic balance of the adults, the food requirements for maintenance, requirements of young, and the weather constraints both as a limit to foraging success and as increased energetic costs for cold weather. The initiation of breeding depends upon firstly finding a suitable territory, and secondly, upon vegetation structure being suitable for nesting. Breeding success depends on the habitat being able to fulfil the energetic requirements of the birds during the breeding period and the survival of eggs and nestling. This is determined by food resource quantity and availability, and is a function of management, weather and skylark behaviour. Birds may also be disturbed during nesting e.g. by farming activities, but this is rare during the breeding season.

The model has been extensively tested and is capable of reproducing a full range of real world skylark population and individual behaviours. These include the mean and variation around time to hatch and nest leaving, densities of skylarks per farm, and within season phenology under different field conditions (Topping, Odderskaer et al. 2013).

8.4.2.2.1 Pesticide Effects

Exposure to the pesticide is calculated based on the amount of pesticide ingested as a result of eating contaminated food. All insect food in the model 'knows' whether it comes from a contaminated area, and if so what the concentration of pesticide per unit of food is. Assuming a skylark eats only contaminated food at X1 application rate it gets 0.005 mg per kcal eaten. However, the kcal eaten depends on energy usage this is varies with activity and temperature. Therefore if we assume it is warm and there are no flight costs, then the intake will be 18kcal = 0.09mg (based on energetic calculations in the skylark model (Topping and Odderskaer 2004)). Adult skylark weight is typically 38g, hence body burden per mg/g = 0.00237 (i.e. 0.09/38). Further assuming we have a 7 day DT50, a trigger value of 0.002 would have an effect only on the day of spraying.

Therefore, the trigger effect for one day after spraying should be set to 0.002 will have an effect only on one day after spraying. A value of 0.001 will have an effect for 7 days following spraying. However, this assumes that the skylark feeds exclusively on contaminated food from within the directly over-sprayed area. If the skylark does not eat 100% from contaminated areas then the chance of triggering this decreases very fast. Since in most cases the skylark will not feed in this way, this is not reasonable for all skylarks. Consequently, a value of 0.001 was used for all scenarios to catch effects that might be caused by drift into off-crop areas, and be comparable to the effects calculated for the other species.

8.4.3 Bembidion lampros (Beetle)

8.4.3.1 Species description

Bembidion lampros Herbst, is a spring breeding ground beetle (Carabidae) that occupies temperate agricultural landscapes. It represents a numerically large group of carabid beetles with common life histories. It is a Palearctic species that has been studied intensively as one of the most common beetles in European agroecosystems, and therefore has a well-described biology and natural history.

Bembidion behaviour is characterised by annual dispersal and aggregation phases with aggregation linked to non-cultivated habitats and dispersal and breeding largely occurring in open areas.

8.4.3.2 Model overview

The *Bembidion* model's individuals are agents designed to simulate the ecology and behaviour of individual beetles. Due to the very high number of beetles in the real world we use the super-individual concept (Scheffer, Baveco et al. 1995), using each beetle agent to represent 100 real-world beetles. Since the environment is dynamic, the resultant response of the sum of the agents' interactions with each other and their environment, through space and time, produces an emergent population response. The original model was described in Bilde and Topping (2004), and full documentation is available in ODdox format (Topping 2009).

Primary drivers in the model are temperature-controlled developmental rates of eggs, larvae and pupae, together with adult beetle interactions with the landscape, and larval interaction: Interactions between beetles are limited to cannibalism by larvae and dispersal or aggregation triggers:

- Larvae exert the major density-dependent upper-limit control via cannibalism. Each super-individual larva reacts to the local environmental drivers of beetle density within a 2-m radius.
- All beetles react to global weather drivers for development and adults also use weather queues for reproduction triggered by weather dependent dispersal. All non-adult stages use day-degree models for controlling development.
- Landscape management, primarily agricultural practices, affect beetles directly, e.g. ploughing causes direct mortality (Thorbek and Bilde 2004).

8.4.3.3 Pesticide Effects

The response to the pesticide is built into the model by assuming a threshold concentration above which there is a daily probability of mortality. This probability (*p*) is calculated from $(1-m) = (1-p)^d$, where *m* is the proportion assumed to die (e.g. 0.8 for 80% mortality over the test period) and *d* is the number of days over which the test was carried out. If the beetle finds itself in a 1-m^2 grid cell with an environmental concentration above the trigger, then it is assumed to die with probability *p*. Note there is no dose-response, so the maximum death rate is set as *m* over *d* days.

8.4.4 Great Crested Newt

Foto Rainer Theuer

8.4.4.1 Species description

The great crested newt is widespread European species it's range extends from Great Britain and Brittany in the west, across much of Europe north of the Alps, and southeast to the Black

Sea. In Denmark, it breeds in medium to small ponds where there are no fish. The female lays eggs individually or in very small groups during the breeding season. The eggs hatch to produce larvae that are predatory and grow until they undergo metamorphosis into ‘efts’ i.e. terrestrial, air-breathing juveniles, at about 4 months of age. These juveniles then live outside the pond for up two years before returning as breeding adults. They are generally philopatric, but will sometimes breed in ponds that they have experienced during their dispersal phase as juveniles or adults.

Exposure of this species to pesticides in agricultural situation can be high. For instance, as observed by Cooke (1986) with adult great crested newts preferring mature wheat fields to marshland or woodlands. In addition, the ponds in which they breed are often located in or by agricultural fields, bringing all stages into potential exposure to pesticides. Listed on Annex IV of the habitats directive, which means that this species is subject to a strict protection regime applied across their entire natural range within the EU, both within and outside Natura 2000 sites. It has been suggested that this species is a good indicator of botanical diversity in the ponds in which it breeds (Gustafson, Pettersson et al. 2006).

8.5 Development of the *Triturus cristatus* model

Development of the newt model followed a pattern-oriented modelling approach used to develop previous ALMaSS models. Pattern-oriented modelling (POM) refers to the multi-criteria design, selection, and calibration of models of complex systems (Grimm and Railsback 2011). The basic idea of POM corresponds to the overall strategy of science, i.e. to use observed patterns, which are characteristic of a certain system, for detecting the mechanisms that generate these patterns and therefore are likely to be key elements of the system’s internal organization (Grimm, Revilla et al. 2005). For complex systems, single patterns are usually not sufficient to narrow down the range of possible generative mechanisms. Therefore, multiple patterns observed at different scales and hierarchical levels are used. For example, cycles in the abundance of small mammals are a striking pattern, but usually do not contain enough information to unambiguously identify the mechanism that generates these cycles in reality. Additional patterns are needed, for example changes of cycle characteristics in response to weather, latitude, type of predators, etc., or changes in behaviour in high- and low-density situations. POM is used implicitly by many experienced modellers, but it has been suggested that it be made an explicit strategy for utilizing observed patterns in a more systematic way (Grimm, Frank et al. 1996, Wiegand, Jeltsch et al. 2003, Grimm and Railsback 2005, Grimm, Revilla et al. 2005, Railsback and Grimm 2012).

In order to carry out the POM procedure the real world patterns needed to be identified. In addition, basic information about the newt, its biology and behaviour, needed to be collected. Hence, the initial steps involved a literature survey for papers related to *Triturus* spp. A Web Of Science search returns over 2300 articles on *Triturus*, but after exclusion of molecular and physiological articles this was reduced to 999, of which 25% mentioned *T. cristatus.* These papers formed the basis from which the parameters and data patterns were extracted.

8.5.1 Fitting

Surprisingly there were very few useful literature patterns for fitting the newt model. Hence the majority of patterns used are weak patterns (Wiegand, Jeltsch et al. 2003), which means that they are indicative of the system behaviour without being able to be precisely quantified. Unfortunately, many of these weak patterns are a direct outcome of the parameters and mechanisms included in the newt model, for example the seasonality of development, migration and reproduction. These should not therefore be used to assess model performance. Of the remaining weak patterns the only one that was used to inform model choices was that it should be possible for spatial dynamics to cause large fluctuations and even occasional local extinctions in a connected meta-population (Griffiths and Williams 2001).

Patterns used for fitting were:

a) The mean long-term density of adults under ideal conditions. This was calculated as 827 adults in a pond of 400m^2, including surrounding vegetation. We made the following assumptions i) that surrounding vegetation occupies 10% of the area; ii) that newt density is constant with pond size up to a ceiling pond size of 400m^2. Hence, ponds above this size are considered to be 400m^2. The highest recorded reliable density appears to be from Karlsson, Betzholtz et al. (2007) who used an intensive drift net sampling to estimate newt numbers in Swedish pond of 100-500m2, which we assume was 300m^2 (the mid-point in the estimation). Target was therefore a density of 83 newts per 100m^2

b) Annual survival statistics. Adult survival was based on Griffiths, Sewell et al. (2010) who estimated inter-annual adult survival over a 12-year period in Canterbury, Kent, UK. Survival ranges over this period for adults were 25-80%. Survival from juvenile to adult is difficult to estimate. The only estimate available comes from (Hedlund 1990) who found that 0.5% of all offspring were recruited to the breeding population. This combined with an estimate that egg to metamorphosis mortality is 95% (Arntzen and Teunis 1993), indicates that mortality of the juveniles must be approximately 90%, but spread over an average two-year development. Targets were therefore that newt survival was 25-80% for adults, 10% for juveniles to adults, and 5% for egg to juvenile.

c) Population fluctuations. Both Arntzen and Teunis (1993) and Griffiths, Sewell et al. (2010) show large inter-annual fluctuations in newt population sizes of over 400% of the lowest population size. Although (Arntzen and Teunis 1993) started from newly created pond population peaked at 350 individuals falling to 16 two years later.

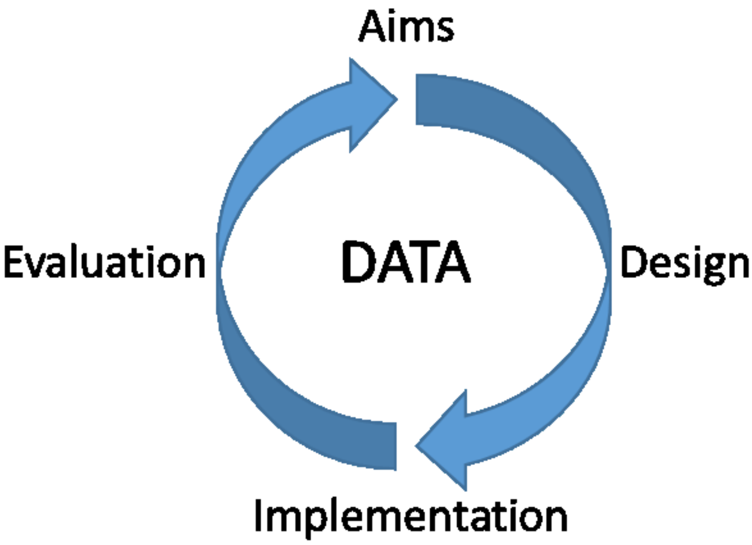


Figure 55: The modelling cycle as implemented in the newt model development. The cycle is fed by data.

8.5.1.1 Fitting process

The development process used in POM is iterative. The process of evaluation from Figure 55 was based on the data patterns from the real world, i.e. those patterns ‘a-c’ described above. The cycle stops when the targets in a-c are sufficiently replicated. The model evaluation step was also structured into iterative steps, similar though simpler than used to develop the ALMaSS vole model (Topping, Dalkvist et al. 2012).

The steps were:

1) Fitting density to a map containing four 400m^2 ponds to pattern ‘a’. Four ponds were needed and placed 200m apart because of the likelihood for extinction of a single pond.

Four ponds increased the overall population survival considerably allowing statistics to be obtained more easily;
2) Evaluating the survival statistics (b) for a 10 x 10 km landscape supporting average newt densities (Esbjerg landscape was used);
3) Evaluating the level of fluctuations in population size;
4) Return to ‘1’ and re-check that changes did not alter densities.
5) If all fits are acceptable stop the cycle, otherwise loop through again.

The evaluation cycle is therefore also iterative but within the overall model cycle and is limited to fitting using parameter value changes. If it seemed impossible to fit the model using this process then the aims, design and implementation need to be revisited to alter the model by including or changing mechanisms.

8.5.1.2 Final model fits to patterns

The overall fitting process is a long one with many iterations and model changes. Here we describe the final four iterations following a point at which the model was originally considered fully functional.

Iteration 1:
In the four-pond landscape, which must be assumed to be perfect and most stable conditions the population fluctuated greatly for the best fit parameter set-up (Figure 56), which required very long simulation runs to provide stable statistics. 1000-year runs were used and those where the population became extinct were discarded. This pattern was fitted using weather data from Canterbury, UK for the same period as the pattern data (1994-2006). Generally, populations of adults were relatively stable at a low level, with regular medium-term peaks of population. Maximum population size was over 1200 in a 1000 year run, but typical population peaks were between 4000 and 6000, and mean adult population size of 1454.

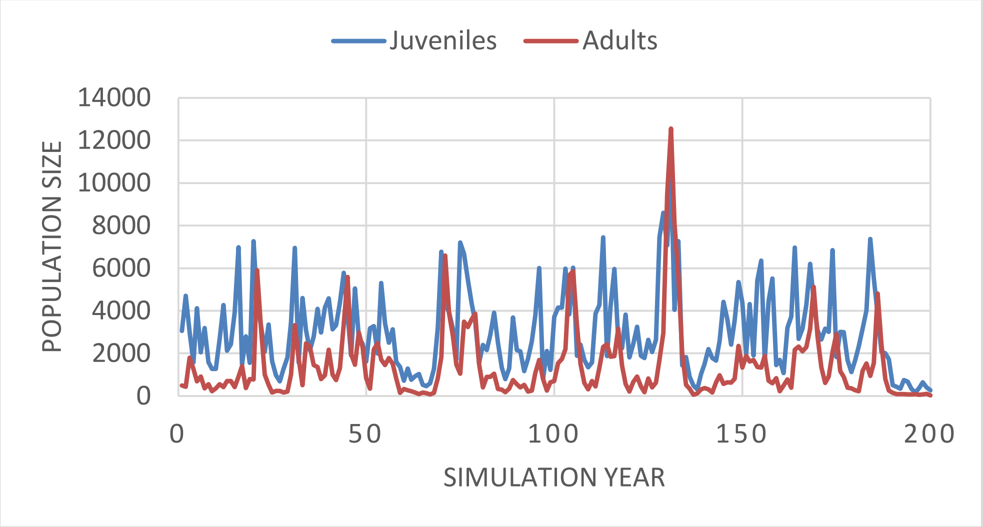


Figure 56: The four-pond landscape showing adult and juvenile numbers over the first 200 years of a 1000-year simulation for iteration 1.

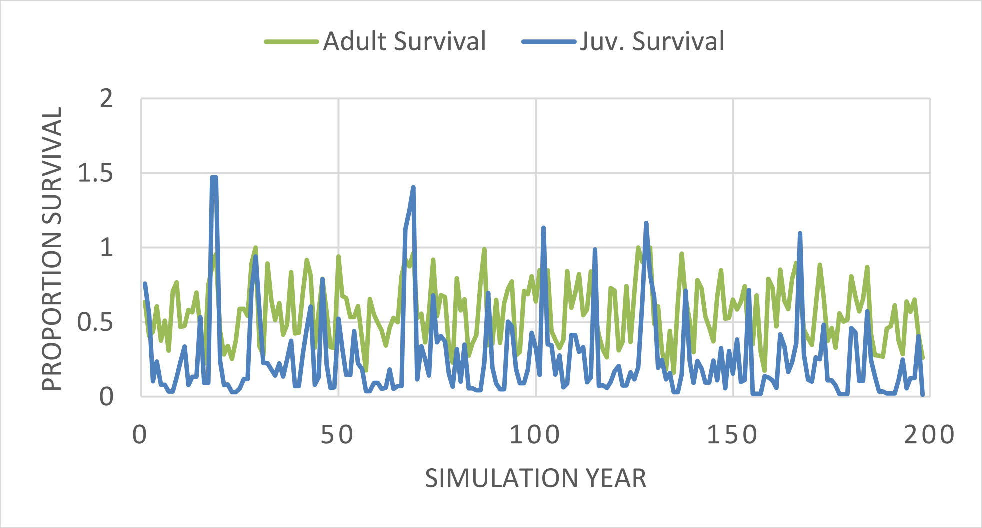


Figure 57: The four-pond landscape showing adult and juvenile survival over the first 200 years of a 1000-year simulation for iteration 1.

The to-adult survival was complicated by the fact that development and survival is over more than one year (so in some years, lower maturation occurs due to weather, but these individuals mature in the following year). Hence, maturation can be over 100% due to delayed maturation from a previous year (Figure 57).

Another choice was presented in this data. The assumption of 5% survival from egg to eft is based on a single study over a relatively short period. Hence, should the target be the overall survival or the mean annual survival? Due to the skewed distribution of inter-annual survival these two figures are rather far apart. The decision was taken to use the overall long-term statistic, not least because this is not biased by variations in to-adult survival time.

Iteration 2:

Some targets proved impossible to fit simultaneously. To-adult survival of 10% meant it was not possible to maintain the population for more than a few decades unless adult survival was increased to >90%. Since the data supporting the adult survival is much stronger (based on Griffiths, Sewell et al. (2010)) than the inferred survival of efts to adult we decided to aim for the published adult survival values and to fit the lowest eft survival commensurate with long-term population persistence (changing the aims in Figure 55). The initially accepted best fit for the four-pond landscape is shown in Table 48. Mortality parameters to create this fit were based on Griffiths, Sewell et al. (2010) but needed to be increased in severity by 2.5% to reduce mean adult survival.

	Survival		
	To Juv	To Adult	Adult
Mean	10.1%	30.4%	57.8%
Max	18.5%	2300.0%	100.0%
Min	0.3%	0.1%	10.8%
Overall	4.9%	25.1%	

Table 48: A set of results from the initially accepted parameter set for the four-pond landscape. Values > 100% are caused by delayed recruitment. Overall survivorship is the total survivorship of the newts to that stage over the course of the whole 1000-year simulation.

Iteration 3:

The iteration 2 patterns of survival were not considered to be satisfactory, in particular juvenile survival appeared to be too variable, based on a magnified adult effect. Therefore, rather than use the same adult survival but increased by a factor (e.g. 1.5 was used in the iteration 2 example), juvenile daily mortality was increased by addition of a fixed daily probability of death (of 0.1%). This reduced fluctuations markedly, and seemingly removed the effect of longer-developmental time, reducing almost all survival percentages to less than 100% (although these effects are still present but hidden in the data) (Figure 58). During this iteration, the translocation of the model from the UK testing data to Danish conditions was tested. The effect of changing weather from Canterbury 1994-2006 to realistic weather for Denmark during the last 30 years (based on Mid-Jutland 1984-2014) resulted in extinction of the population under the optimal 4-pond landscape. Since newts are not extinct, this indicates a fault in the model when translocating it from Canterbury to Denmark (i.e. the newts in Denmark do not respond to weather in the same way as those in the Canterbury study). Although it should be borne in mind that the Canterbury population was considered in decline. After considerable testing it was found to be impossible to reach all goals simultaneously (survival % could be fitted well, but long-term population survival could not), and final round of the model cycle was initiated.

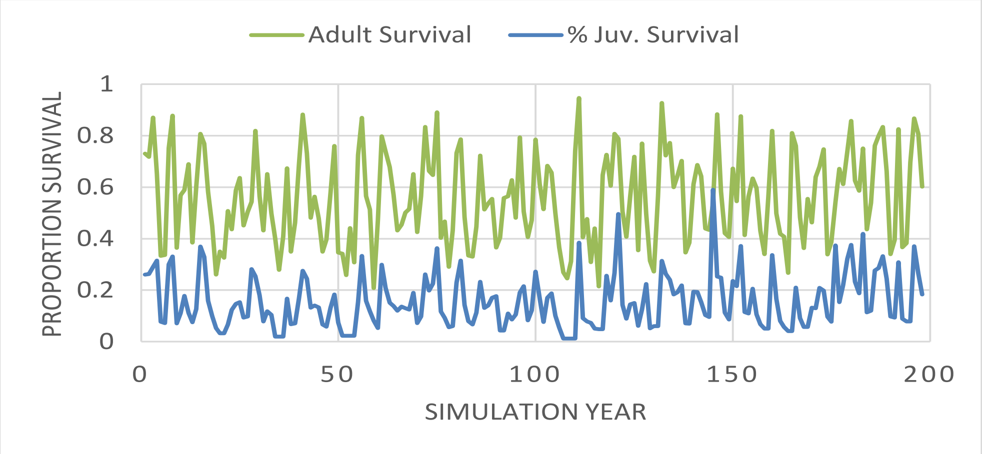


Figure 58: The four-pond landscape showing the effect of adding a fixed daily probability of death on adult and juvenile survival over the first 200 years of a 1000-year simulation for iteration 3

Iteration 4:

In order to ensure population survival over a longer period the larval food competition was altered from the initial situation where all larvae ate food until it was gone (pure scramble competition), leading to death if there was not food enough, to a situation where the last 1% of food remains. This effectively provides a scramble competition situation for 99% of the food, then contest for the last 1%. This prevented the total extinction of larvae in a pond due to food shortage, but allowed only very small numbers to survive under these conditions.

The resulting fit to the target patterns was good under Danish conditions. Fluctuations in adult population size were still large, but not as large as the in iteration 1 (Figure 59), but easily satisfying the 400% variation required by the aims. Population density was fitted to the target and this resulted in good survival fits (Table 49). Adult survival ranges were also a good fit to the targets (Figure 60 & Table 49).

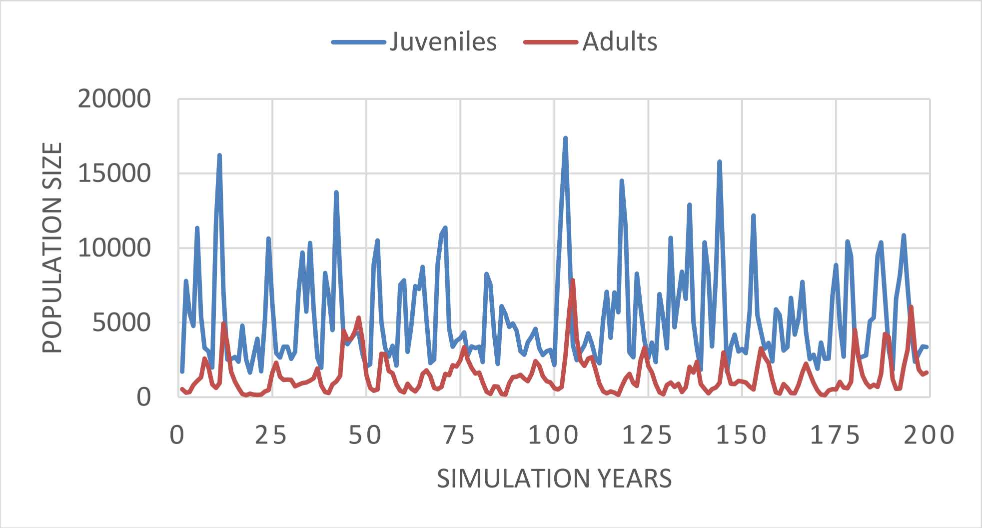


Figure 59: Fluctuations in numbers of adults and juveniles from year-to-year in the first 200 years of a 1000-year run of the final parameter set with Danish weather.

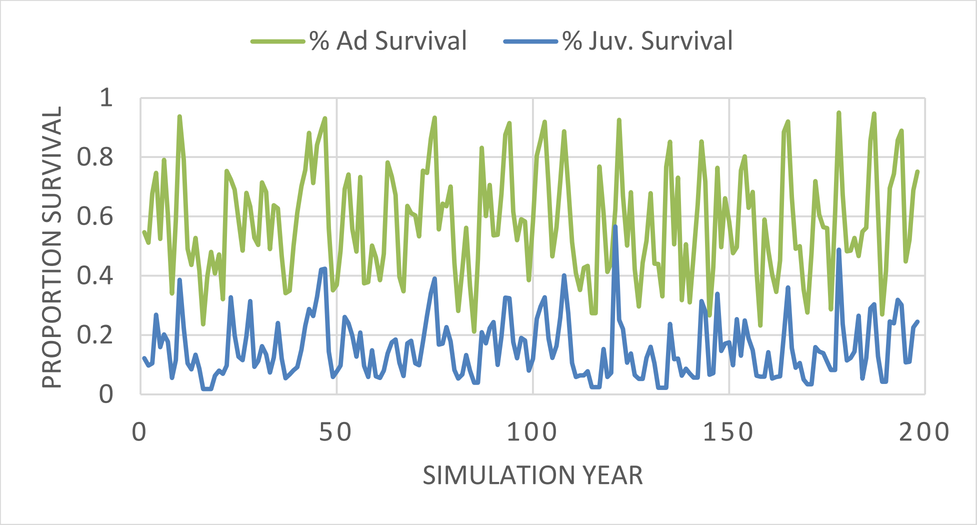


Figure 60: Fluctuations in survival of adults and juveniles from year-to-year in the first 200 years of a 1000-year run of the final parameter set with Danish weather.

	Survival		
	To Juv	To Adult	Adult
Mean	11.6%	14.7%	56.8%
Max	27.6%	56.6%	98.1%
Min	0.5%	1.6%	17.7%
Overall	5.1%	15%	

Table 49: A set of survival results from the final parameter set with Danish weather for the four-pond landscape. Overall survivorship is the total survivorship of the newts to that stage over the course of the whole 1000-year simulation.

Re-running the final fit on the four-pond landscape with Canterbury weather resulted in a poorer fit to patterns than previously. Mean population size was increased from 83 to 118 newts per pond. Adult survival remained in the same range, but juvenile survival increased from 15 to 26% and to juvenile survival decreased from 5% to 2.6%.

Running the model on realistic landscapes indicated that landscape has a large influence on these fits. For example based on 1000-year long simulation in Karup to-juvenile survival was 6%, to-adult 14% and mean adult survival was 57%, whereas on Esbjerg the figures were 2.7%, 23% and 57%, and for Næstved 7.3%, 12% and 56%.

8.5.2 Resulting model overview

Please note that a more readable and navigable version of this description is available in the ODdox (https://almassdocs.au.dk/ALMaSSODdox/Newt/index.html), which represents the full model documentation including code.

The newt model considers newts developing from eggs to adults, moving, reproducing and dying. Newts are represented by five life-stages, namely eggs, larvae, free-living juveniles, adult males and adult females. Each individual is represented uniquely in the model, and each newt at any point in its modelled life is a member of one of the five life-stages. In model parlance, each is an instance (an object) of the object class representing its life-stage. When an individual changes life-stage e.g. an egg hatches, then a new instance of the new life-stage class is created and all information copied from the old object to the new one before destroying the old object.

Each newt object has unique characteristics e.g. its age and location, and can carry out behaviours associated with its life-stage. Following the ALMaSS paradigm these behaviours are represented as a state-machine, i.e. a newt is in a behavioural state until an internal or external change triggers it to change to a new behaviour.

To minimise the code size and improve code readability and maintenance object oriented principles have been applied to many aspects of code design. Therefore, although all newt objects must be members of life-stage classes, there are also some special life-stage classes created to collect attributes and behaviours common to other classes, these are Newt_Base (contains attributes and behaviours common to all newts) and Newt_Adult (contains attributes and behaviours common to adult males and females). These are model constructs and do not represent real-world organisation, however, they appear in the description below and thus need to be mentioned here to avoid confusion.

8.5.2.1 State variables

A list of state variables held by each life-stage class is listed below. These variables contain all the information held internally for each newt object. Note that this list is shorter than in the ODdox since here we have removed all model constructs, leaving only ‘natural’ state variables that can be related to real-world structure or actions.

8.5.2.1.1 Newt_Base

Newt_Base collects all the behaviours and attributes common to all newt life-stages. It contains the following attributes:

- m_Age The age of the Newt (units days).
- m_CurrentNewtState This is the current behavioural state.
- m_pondlist This is the list of pond locations found by the newt, the first value being the pond of birth. Each entry is an index to the list of polygons held by the Landscape class (see interconnections).

- m_body_burden This is the current body-burden of pesticide, it is total amount.
- m_reproductiveinhibition This is a flag denoting reproductive inhibition if true. This can be set by descendent class responses to pesticides typically during developmental stages.

8.5.2.1.2 Class Newt_Egg

Class Newt_Egg represents the egg stage of the newt in ponds. In addition to inherited attributes from Newt_Base, each egg has:

- m_AgeDegrees This is an internal variable used to sum up the day-degrees experienced by the egg/larva in degree days (units degree days).

8.5.2.1.3 Class Newt_Larva

Class Newt_Larva represents the larval stage of the newt in ponds. Each larva has a location in a pond polygon represented by it's home pond polygon reference (first pond in m_pondlist). The larva starts with zero day degrees and inherits the egg age at hatch. It ages each day (see st_Develop), and the temperature it experiences is summed each day to create a cumulative day-degree sum. When this sum reaches a pre-defined number of day-degrees the larva metamorphoses to be a juvenile. Death can occur by daily mortality probability or due to failing to get daily food intake. In addition to inherited attributes from Newt_Egg, each larva has:

- m_LarvalFoodProportion Which holds the value for the proportion of food eaten per day relative to weight.
- m_LarvaDevelopmentUpperSz Holds an input parameter - it is the upper size at which larvae will undergo metamorphosis.
- m_LarvaDevelopmentLowerSz Holds an input parameter - it is the lower size below which larvae cannot undergo metamorphosis.
- m_LarvaDevelopmentTime Holds an input parameter - it is the time before a larva can undergo metamorphosis.
- m_NewtLarvaDailyGrowthIncrement Holds an input parameter - it is the daily growth increment if a larva survives to grow that day.
- m_LarvaMortalityChance Holds an input parameter - it is the daily probability of death from unspecified causes for a larva.
-

8.5.2.1.4 Class Newt_Juvenile

Class Newt_Juvenile represents the juvenile stage of the newt. It is free living and moves in the landscape.
In addition to inherited attributes from Newt_Base, each juvenile has:

- m_weight which is the weight of the newt in g.
- m_CurrentHabitat - the current habitat type the newt is located in.
- m_InPond is the polyrefindex for the pond the newt is in, or -1 if not in a pond.
- m_SimW The width of the simulation map, stored for fast access.
- m_SimH The height of the simulation map, stored for fast access.
- m_OurVector The last direction the newt moved in.

Other new attributes are included here for efficient storage of input parameters:

- m_JuvenileDailyWeightGain Used in determining daily growth, this is the daily increment.
- m_roadmortalityprob The probability of death when crossing a road, this is per movement (1 x m_newtwalkspeed).
- m_newtwalkspeed The max walking speed of a newt per movement (1 day)
- m_newtwalkstepsize The size of a step when evaluating habitat during walking - used to speed up movement if necessary, default is 1m
- m_goodhabitatdispersalprob Probability of dispersal in good habitat

- m_poorhabitatdispersalprob Probability of dispersal in poor habitat
- m_NewtDormancyTemperature Temperature in degrees that the newts become dormant and stop moving or breeding (summed over previous 5 days). Default value is 22.5 based on (Langton, Beckett et al. 2001) who states 4-5 degrees.

8.5.2.1.5 Class Newt_Adult

The adult class handles those attributes and functions common to all adults.
In addition to inherited attributes from Newt_Juvenile, each adult has the following attributes:

- m_AdultLifespan Parameter which is the maximum lifespan of a newt. Set at 14 years following Francillonvieillot, Arntzen et al. (1990).
- m_targetpondx which is the x-coordinate of the pond the newt decides to head to in migration
- m_targetpondy which is the y-coordinate of the pond the newt decides to head to in migration

8.5.2.1.6 Class Newt_Male

Newt_Male adds no new attributes to Newt_Adult.

8.5.2.1.7 Class Newt_Female

In addition to inherited attributes from Newt_Adult, each female has:

m_eggproductionvolume Holds the max number of eggs that can be produced during breeding
m_eggdailyproductionvolume Holds the number of eggs produced per day during breeding
m_eggsproduced is used to keep track of the actual number of eggs produced
m_mated which is used to record whether the newt is mated or not this season

8.5.2.2 Inter-connections

8.5.2.2.1 Landscape

The landscape class is the model construct that handles the environment in which the newts find themselves. This is described in the general ALMaSS description (Section 2), however, there is some functionality related to ponds that is specific to the newt model.

Ponds are represented by the class Pond, which is derived from another ALMaSS class (Element) that describes the basic behaviour of all habitat (e.g. contains its size, vegetation properties, manager if farmed etc.). The pond extends this class with the following attributes:

- m_LarvalFood – this is the amount of larval food present and available for newt larvae.
- m_LarvalFoodScaler – this is the proportion of larval food out of the possible larval food per unit volume.
- m_PondPesticide – this is the pesticide content per cubic metre of water.
- m_MaleNewtPresent – a true/false flag for the presence of an adult male newt.
- m_PondQuality - a factor used to alter the pond qualities (default behaviour is random 0.0-1.0).

8.5.2.2.2 Ponds

The Pond class carries out three behaviours in addition to providing access to its data:

- CalcPondPesticide - Calculates the amount of pesticide per unit pond water. The pesticide is calculated based on the mean concentration per m^2, which is then multiplied by a factor representing run-off from the surroundings. This method assumes a uniform depth of water of 1m.
- CalcLarvalFood – This, together with SubtractLarvalFood below, comprise the density-dependent mechanisms in the newt model, therefore are what ultimately limits the population size. Calculates the amount of larval food present at the beginning of the day. The larval food is calculated assuming a logistic equation in the form of $N_{t+1} = N_t+(N*r * (1-N/K))$ where t is one day, N is a scaler which is multiplied by a constant and the area of the pond to get the

total larval food, K & r are carrying capacity and instantaneous reproductive rate respectively. K can change with season and this is currently hard-coded as 1.0 (i.e. no seasonal variation), but could be an input variable later. The values are held in LarvalFoodMonthlyK. The steps in the calculation are:

  - Enforce an assumed pond size for newts as maximum 400 m^2.
  - Back calculate the current scaler value. This is needed because between time steps, food may be eaten by larvae. This is done based on the area and a fixed constant parameter LARVA_FOODBIOMASSCONST
  - Ensure we never get zero larval food, so there is always something to grow the curve from by restricting m_LarvalFoodScaler to be >=0.01.
  - Calculate the new scaler based on the logistic equation as described above using the input parameter LARVA_FOODBIOMASSR as parameter *r*.
  - Re-calculate the new total food biomass based on the area and a fixed constant held in LARVA_FOODBIOMASSCONST by multiplication.

- SubtractLarvalFood – This function is called by a larva when feeding, removes an age specific amount of larval food. If there is enough food present then the larva will grow, if not the larva is assumed to starve.

8.5.2.2.3 Class Newt_Population_Manager

The Newt population manager is responsible for handling all lists of newts and scheduling their behaviour.
It provides the facilities for output and handles central newt functions needed to interact with the landscape.

- m_BreedingSeasonFlag Set to 0 at the start of the newt breeding season, then records the number of days after
- m_NewtEgg_DDTempRate An array to hold a precalulated day-degree rate transformation for egg development
- m_NewtMetamorphosisStats A class for holding the statistics on newt metamorphosis development times
- m_NewtEggProdStats A class for holding the statistics on newt egg production
- m_NewtAdultProdStats A class for holding the statistics on newt adult production

The Newt_Population_Manager has a number of specialised newt functions. Excluding purely programmatic constructs these are:

- CreateObjects The method for creating a new individual Newt of any life-stage.
- IsBreedingSeason Returns whether at this moment in time it is breeding season or not as true or false.
- IsMate Returns whether there is a (male) mate present in the pond or not as true or false.
- SetFreeLivingMortChance Calculate the daily background mortality chance for free-living newts based on weather.
- GetEggDDRateTransformation Returns the value from the pre-calculated m_NewtEgg_DDTempRate array to get effective day degrees. The calculation is based on the day-degree calculation where the sum of day degrees needed for development at temperature *T* is given by: $NEWT\_EGG\_DEVELTOTAL*T^{NEWT\_EGG\_DEVELDDPARAMETER}$ based on the data provided by D'Amen, Vignoli et al. (2007)
- RecordAdultProduction Output function. Add a new adult to the statistics record
- RecordEggProduction Output function. Add a new egg production to the statistics record
- RecordMetamorphosis Output function. Add a new metamorphosis time to the stats record
- InitOutputMetamorphosisStats Output function. Initialises output mean and variance for meatamorphosis times this year
- OutputMetamorphosisStats Output function. Output mean and variance for metamorphosis times this year
- SetUnsetBreedingSeason Controls when it is breeding season.

- TheNWordOutputProbe This is a tailored version of a general function producing gridded output to support the calculation of Abundance/Occupancy ratios for the Newt.

8.5.2.3 Development

Egg development - Firstly it is determined whether the egg succumbs to daily mortality using a probability test against a daily probability (input parameter). Next if there is the need to test for pesticide effects then the body-burden is adjusted for yesterdays elimination, then pesticide at the egg's location is determined and any intake stored. If the body-burden is greater than the threshold for effects, then the specific Egg function InternalPesticideHandlingAndResponse is called (see Mortality below) and the impacts determined. If the egg has not died by this point then:

- The age is incremented
- The sum of day degrees is increased by the mean temperature today
- If the total sum of day degrees exceeds the parameter value held in NEWT_EGG_DEVELTOTAL, the egg will hatch and be replaced with a Newt_Larva object.
- If not fully developed the process will be repeated the following day.

Larval Development – Firstly, like all newt life-stages there is a probability test to determine whether the larva dies of unspecified mortality followed by age incrementation. Next, there is an evaluation of the food availability in the pond. If there is not enough food for the larva then it will also die. Then if there is the need to test for pesticide effects then the body-burden is adjusted for yesterday's elimination, the pesticide at this location is determined and any intake stored. If the body-burden is greater than the threshold for effects, then the newt larva-specific Internal-PesticideHandlingAndResponse is called and the impacts determined. If the larva survives then the age is incremented and the larva grows a fixed amount of length.

The current model grows the larva from the starting length of 1mm and maturation occurs when the larva reaches a minimum size.

Juvenile Development – Juveniles also undergo a daily mortality probability test. If they survive then they increment their age and will grow a fixed amount each day assuming the temperature is above a developmental minimum. If it is not above a minimum then only the age is incremented. Once they reach adult size they mature.

Adult Development – The adult ages each day. When created it has an allotted total maximum lifespan of 14 years following (Francillonvieillot, Arntzen et al. 1990). If the newt reaches this age it dies (this is a very rare event).

8.5.2.4 Movement

Movement of newts occurs in the model in juvenile and adult life-stages. There are two types of movement considered. The first is dispersal that occurs when a newt (juvenile and adult) leaves the pond and moves around in the terrestrial environment, the second is a migration to breeding ponds as adults.

Juvenile Dispersal – It is assumed that the newt can only move around when the humidity is high. This is related to the rainfall and temperature of the preceding days. If the humidity is high enough then the newt may move otherwise it is forced to stay where it is. The newt moves around using a guided random walk. This means that movement is random except that it will evaluate each step and will not walk onto habitats considered ‘illegal’ e.g. houses. At each step it evaluates if the habitat is walks onto is legal (if not it does not move), it also checks if it is a road in which case there is a probability of road mortality. Finally, it checks if it is a pond and if so remembers this (if not experienced already). The number of steps is controlled by the parameter NEWT_WALKSPEED and is a uniform random distribution between 0 and this value.

When the newt is leaves the pond this behaviour is altered to remove the random walk element, meaning that the first movement after leaving the pond is always in a straight line away from it. Subsequent movements follow the behaviour above.

Adult dispersal – adult dispersal follows the same pattern as juvenile behaviour except that daily movement distances are reduced to 10% of juvenile dispersal following (Karlsson, Betzholtz et al. 2007).

Pond Migration – A return migration to a pond occurs for adults annually and juveniles when they reach maturity. The migration is triggered by the date of first possible breeding. Once triggered the newt moves towards a target pond used the dispersal method above but as a directed random walk using a vector to the pond. The target pond is determined by the last breeding pond experienced by the newt during dispersal, which is overwhelmingly the natal pond unless ponds are placed close together. Depending where the newt is in the landscape it will take variable amounts of time to reach the pond, and maturing juveniles will typically reach the pond after mature adults.

8.5.2.5 Reproduction

Once breeding adults reach the pond breeding starts immediately assuming both sexes are represented at the pond. If not the newts cannot breed and will wait for others to arrive. Once breeding starts the female newt produces a fixed number of eggs per day until her complement of eggs is produced. Both of these numbers can be altered by input parameters. These newts have a genetic peculiarity resulting in 50% of the eggs being sterile (Macgregor and Horner 1980). Newts were therefore constrained to only produce 50% of the normal complement of eggs but assumed all were fertile. Once all eggs were produced the female leaves the pond, the male only leaving at the end of the breeding season (although these often coincide). If the full complement of eggs is not laid by the end of the breeding season the ‘extra’ eggs are lost.

8.5.2.6 Mortality

Pesticides – Each life-stage has a specific InternalPesticideHandlingAndResponse function that determines how it responds to concentrations of different pesticide types. Both the concentration and response independently adjustable for each life-stage by changing input parameters. In the scenarios run for this report only adult and juvenile direct mortality was used.

Road mortalities – newts walking on roads have a probability of being killed by traffic depending on the type of road and therefore the associated traffic load. This is controlled by the parameter NEWT_ROADMORTALITYPROB. The probability of death when crossing a road, this is per movement, which means that this will increase for larger (wider roads).

8.5.2.7 Model exclusions

Things that were not included but are known to be important:

- Breeding success has been linked to the quality of vegetation around the pond (Vuorio, Heikkinen et al. 2013), with the quality of surrounding resources having been shown to be important in the level of egg production (Halley, Oldham et al. 1996), and amphibian persistence is often linked to the quality of pond-surrounding habitat (e.g. Guerry and Hunter 2002).
- Pond quality determines the success of reproduction, e.g. pH is an important factor. Egg hatching is prevented at pH of less than 4.5 (Griffiths and Dewijer 1994), and also affects larval growth (Griffiths, Dewijer et al. 1993).This is currently included as a stochastic effect.
- Pond hydrology is also an important factor. Ponds drying is thought to be a major contributor to local extinctions and lowered population reproduction (Karlsson, Betzholtz et al. 2007).
- Fish are considered to be a major constraint on newt breeding with little success in ponds containing predatory fish (Hartel, Nemes et al. 2007).

Full documentation in ODdox format is available from
https://almassdocs.au.dk/ALMaSSODdox/Newt/index.html

8.6 Sensitivity analysis of the *Triturus cristatus* model

The newt model is a computationally intensive model, therefore run times are long. This precludes a Monte Carlo-based sensitivity analysis where every combination of possible parameter values are tested. As a result a one-at-a-time sensitivity analysis has been carried out for each parameter.

8.6.1 Sensitivity per parameter

For each parameter there is a short description, a default value and an evaluation of uncertainty on a categorical scale of very low, low, medium, high, and very high. These uncertainty categories refer to the isolated uncertainty of the individual parameter, but see the uncertainty analysis section for a summary and analysis of the way uncertainty propogates through the model.

There is also a sensitivity plot for each parameter showing the relative change in population size by varying the parameter by +/- 15%, 35% and 50% of its default value (in some cases, due to parameter value or format these proportions were changed slightly). Like uncertainty, the sensitivity of the population size to the parameter is categories as very low, low, medium, high, and very high. This categorisation is based on the total range covered by the parameter over +/-50%. It is calculated by the formula (range/2.875)/(2.875/5.0). Results 0.0-0.9999 are very low, low is 1.0-1.9999, medium is 2.0-2.9999 high is 3.0-3.9999 and very high 4.0 or above.

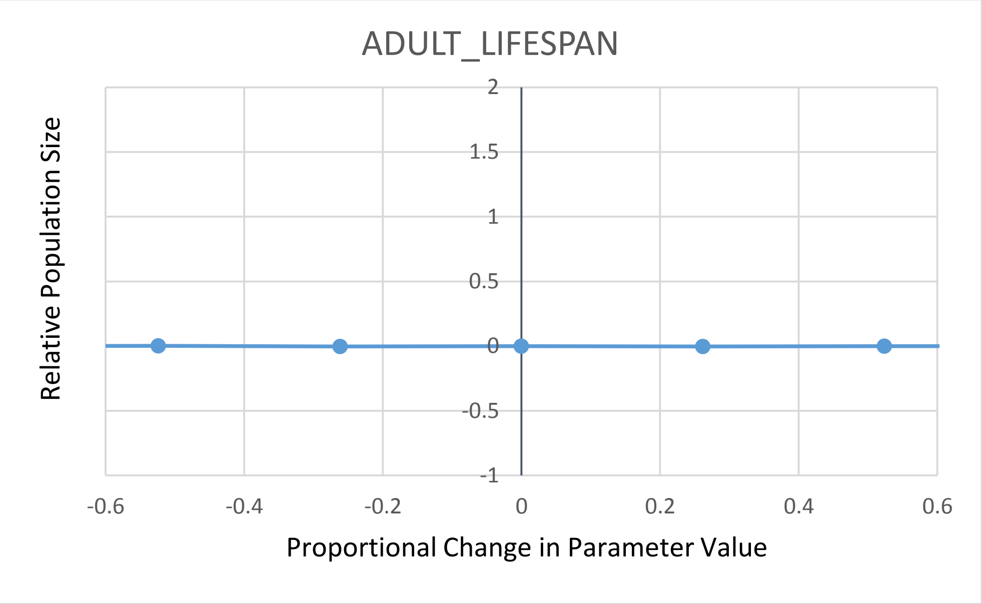
8.6.1.1 ADULT_LIFESPAN

Default value: 5110

Sensitivity Category: very low

Uncertainty category: Low

This parameter is the maximum lifespan in days for an adult (including egg and larval stages) Default value is 5110, or 14 years, (Francillonvieillot, Arntzen et al. 1990). This parameter has virtually no effect on the simulation because the vast majority of newts do not survive anywhere near this length of time. This value comes from maximum observed lifespan in captivity.



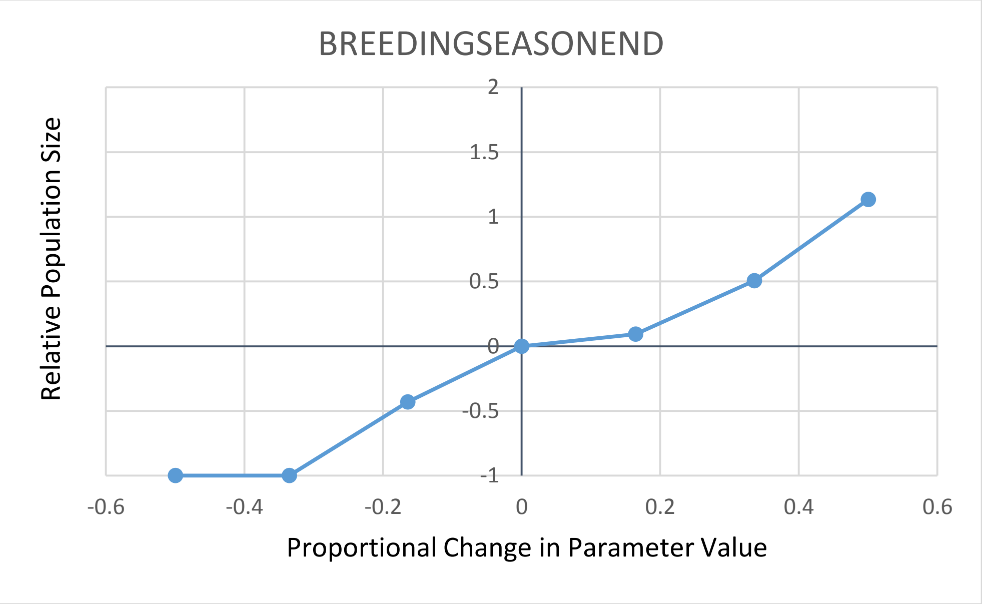
8.6.1.2 BREEDINGSEASONEND

Default value: June 1st

Sensitivity category: Very High

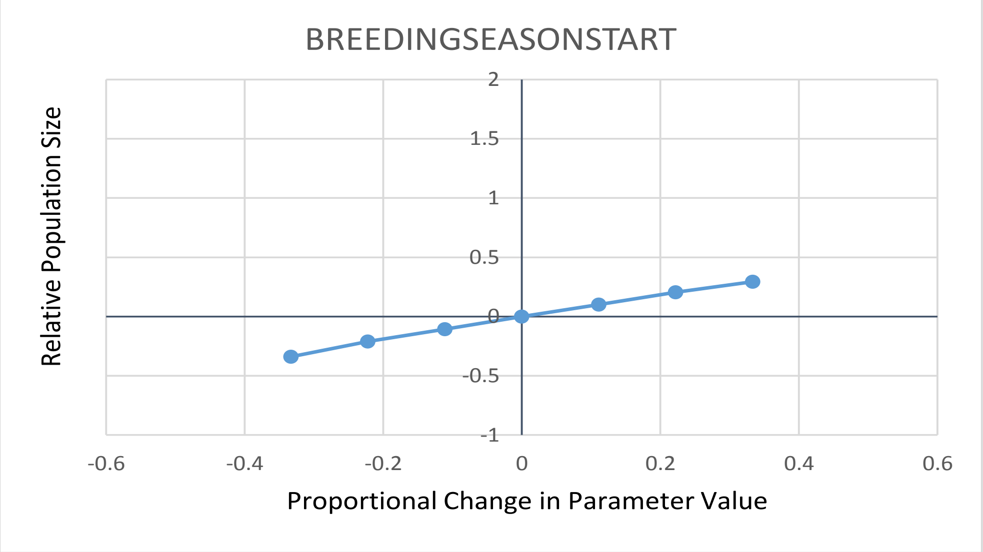
Uncertainty category: Low

Parameter represents the latest possible breeding date. Default value is 1st June (Langton, Beckett et al. 2001, Griffiths, Sewell et al. 2010). Here we use a fixed value, but this is not really the case, the newts emerge from the pond over a period of a month (Langton, Beckett et al. 2001). This is simulated in Newt_Adult::st_EvaluateHabitat by using a daily probability of emergence from pond which is 1 after 30 days.



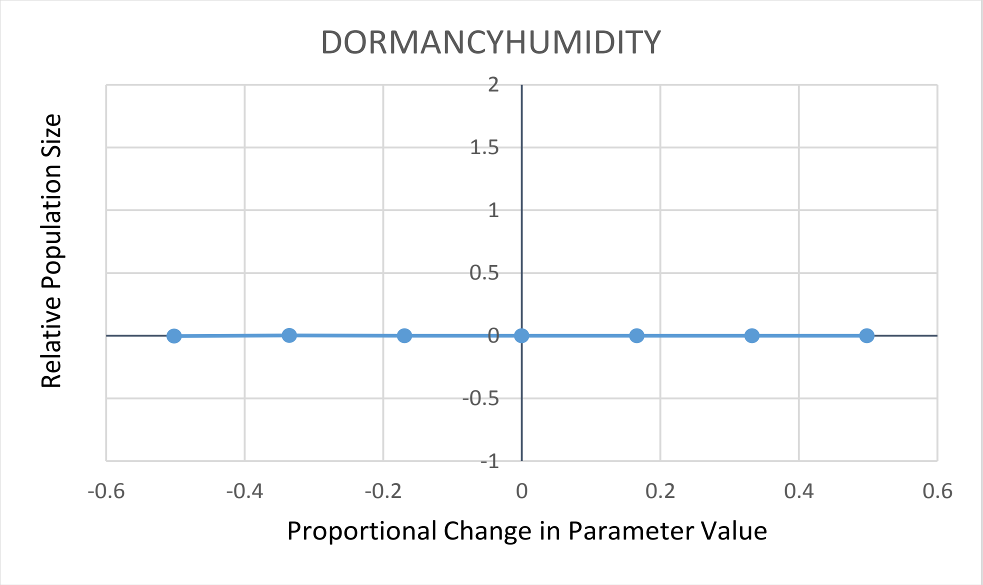
8.6.1.3 BREEDINGSEASONSTART

Default value: March 1st
Sensitivity category: Low
Uncertainty category: Low
Parameter represents the earliest possible start to breeding season. The value is based on Griffiths, Sewell et al. (2010) and Langton, Beckett et al. (2001). Increasing or decreasing the breeding season start has a linear effect on population size, with an increase/decrease of 50% altering the population size by +/- 30%.



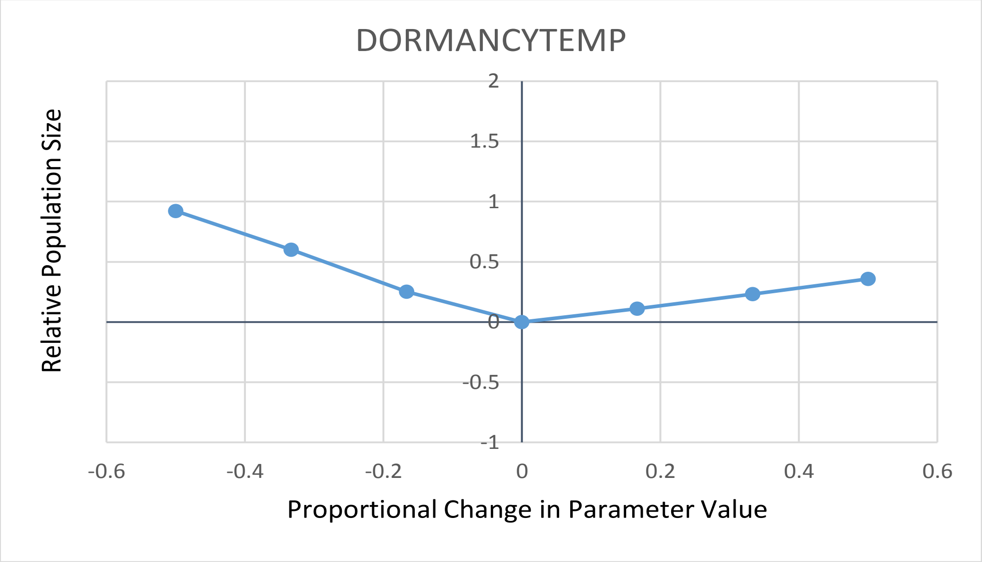
8.6.1.4 DORMANCYHUMIDITY

Default value: 0.003125
Sensitivity category: very low
Uncertainty category: Medium
This parameter is the humidity level needed for activity. Default value of 0.003125 represents 1mm rain 5 days ago at 20 degrees. The simulation is completely insensitive to this parameter due to the fact that the rainfall levels appear to be in excess of this value most of the time, even in summer.



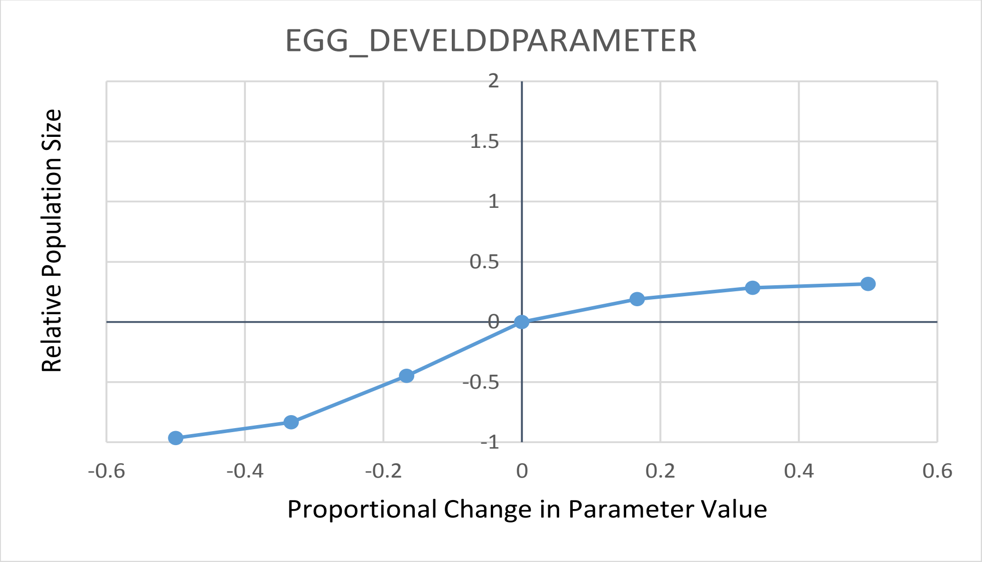
8.6.1.5 DORMANCYTEMP

Default value: 22.5
Sensitivity category: Medium
Uncertainty category: Very Low
This parameter is the temperature needed for newt movement (over last 5 days) Default value is 22.5 based on (Langton, Beckett et al. 2001) who state 4-5 degrees. The response to changes in this parameter are unusual, with the default value giving a population size minima and increases or decreases in the value increasing linearly either side. Decreasing the dormancy temperature threshold increases population size more than increasing the parameter value.



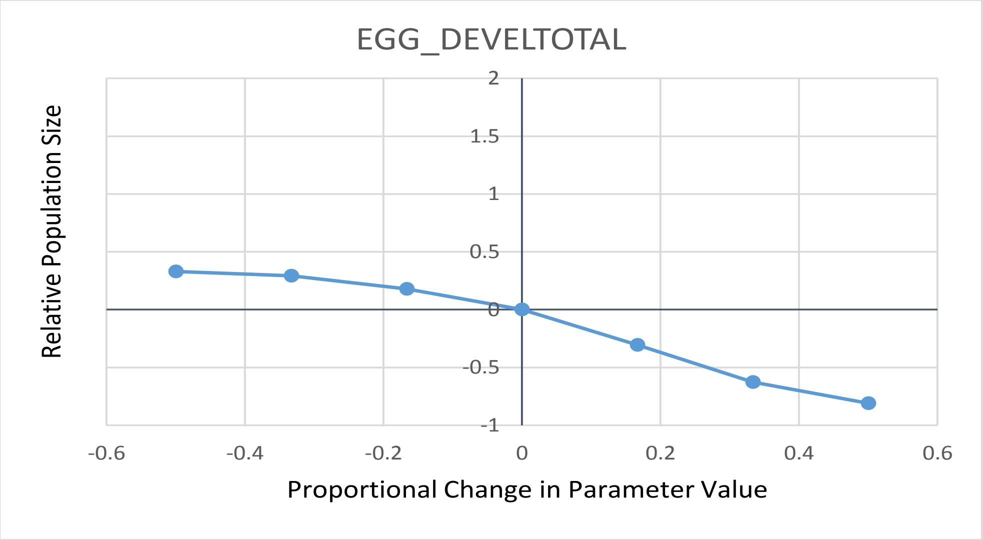
8.6.1.6 EGG_DEVELDDPARAMETER

Default value: -0.534
Sensitivity category: Medium
Uncertainty category: Low
This is the temperature slope for egg development change with temperature. Default value is -0.534 fitted to *T.carnifex* data from D'Amen, Vignoli et al. (2007). Since it is not measured on *T. cristatus* there is a species-specific uncertainty associated with this parameter, but these are morphologically similar species and in the same genus, therefore uncertainty here is relatively small.



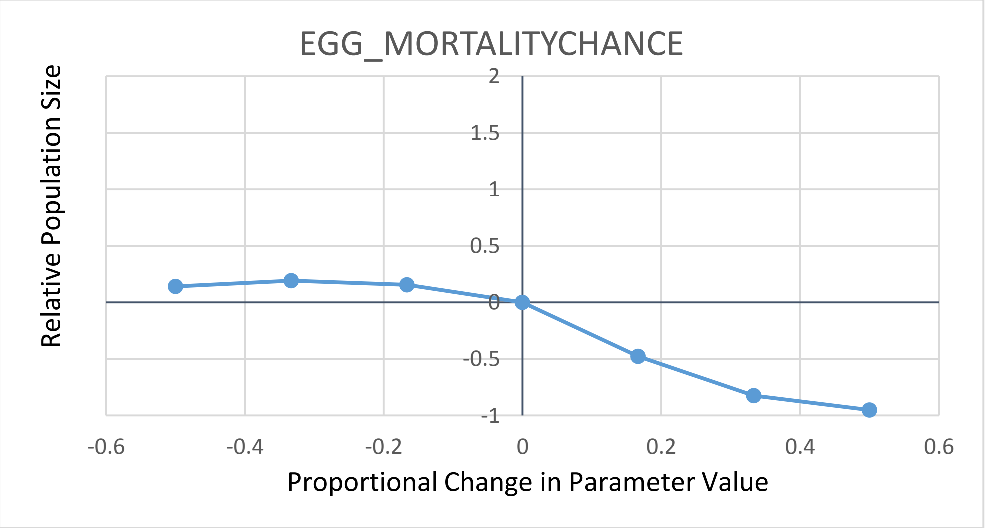
8.6.1.7 EGG_DEVELTOTAL

Default value: 900
Sensitivity category: Medium
Uncertainty category: Very Low
This is the sum of day degrees before an egg hatches. Default value is 1508 fitted to *T. carnifex* data from D'Amen, Vignoli et al. (2007), but then calibrated downwards to obtain developmental rates fitting with *T.cristatus*. Since it is not measured on *T. cristatus* there is a species-specific uncertainty associated with this parameter, but these are morphologically similar species and in the same genus, therefore uncertainty here is small, especially since the value is calibrated to *T. cristatus* data.



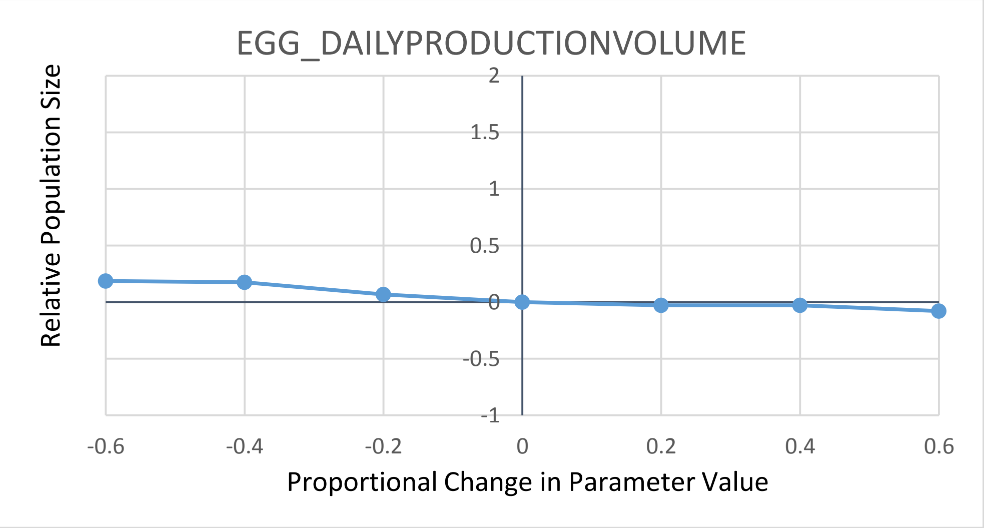
8.6.1.8 EGG_MORTALITYCHANCE

Default value: 0.01
Sensitivity category: Medium
Uncertainty category: Medium
Daily egg mortality probability, excluding environmental mortality. Fitted to the assumption of 95% mortality between embryo and metamorphosis (Hedlund 1990). Parameter uncertainty relates to the variability in overall egg mortality. There is little data on this, but it is unlikely that mortality rates are much higher than 95% in general, so variation is likely to be in the lower end of the parameter range.



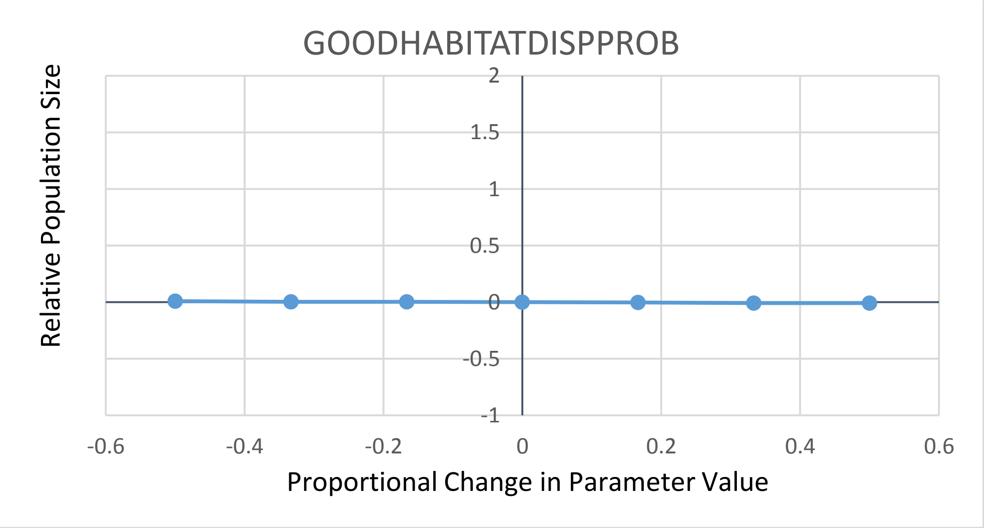
8.6.1.9 EGG_DAILYPRODUCTIONVOLUME

Default value: 5
Sensitivity category: Very Low
Uncertainty category: Low
This parameter represents the number of eggs laid per day when breeding. At the start of the breeding season females may lay just a few eggs per night, but as air and then water temperatures rise, by April they may lay ten or so eggs each day, with consecutively laid eggs often deposited on the same plant (Langton, Beckett et al. 2001). Currently this mechanism is not built into the model and the mean value of 5 is used. Uncertainty is low since this data is based on field observation and is consistent with naturalists reports.



8.6.1.10 GOODHABITATDISPPROB

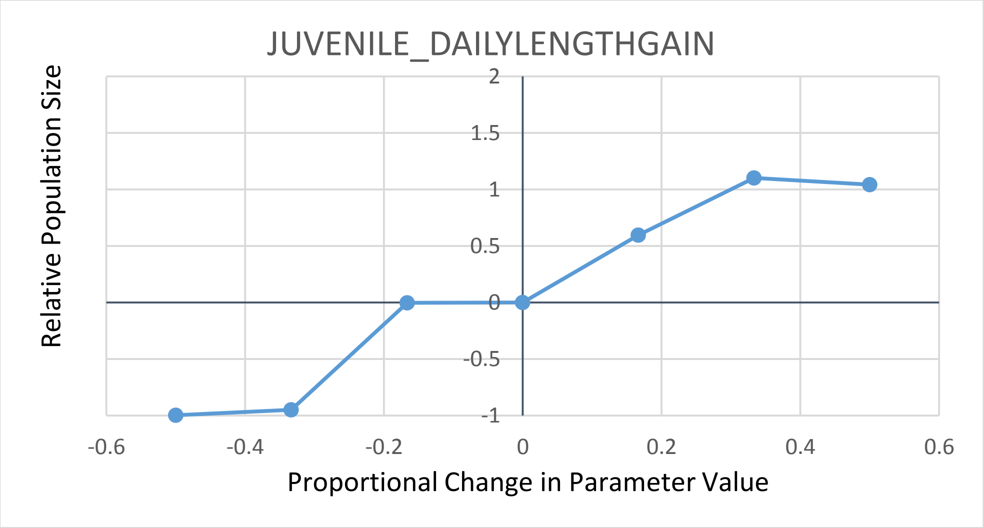
Default value: 0.25
Sensitivity category: Very Low
Uncertainty category: High
This parameter represents the probability of dispersal for free-living newts when in good habitats. It is arbitrarily set to one quarter of the probability of moving if in poor habitat. Increasing dispersal decreases population size, but there is only 1.5% change across the whole range of values tested, indicating that this parameter is of very minor importance.



8.6.1.11 JUVENILE_DAILYLENGTHGAIN

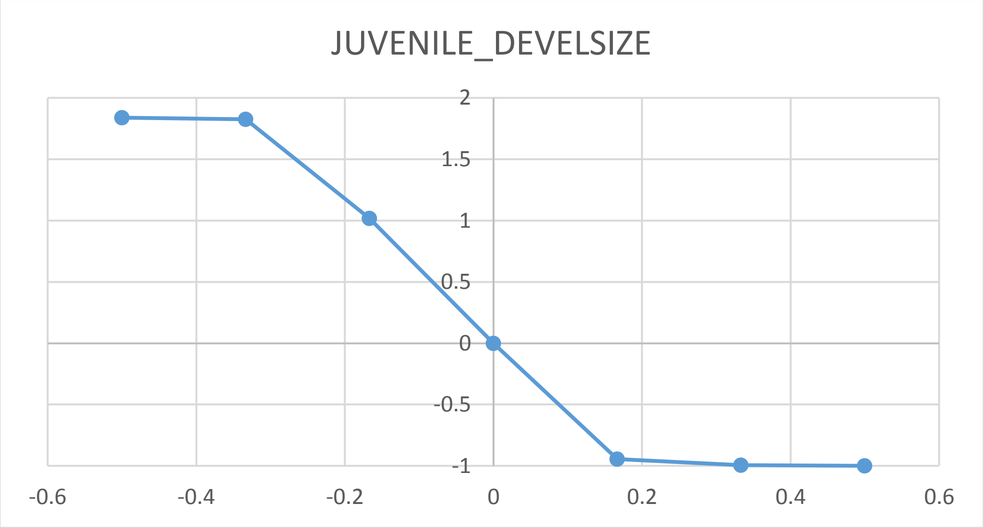
Default value: 0.047827

Sensitivity category: High
Uncertainty category: Low
This parameter controls growth rate of juveniles. If the temperature is above the NEWT_DORMANCYTEMP the newt grows by this many mm per day. Total growth under lab conditions is 21.9mm over 458 days (Baker 1998). The stepped response curve indicates a threshold effect of development spanning zero or more years. Measured in the laboratory, the value of this parameter is quite certain.



8.6.1.12 JUVENILE_DEVELSIZE

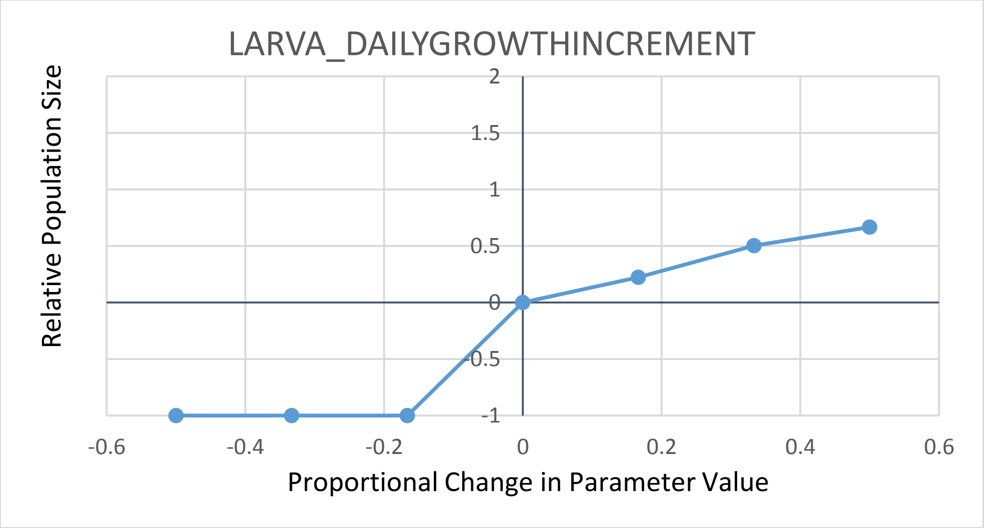
Default value: 63.9mm
Sensitivity category: Very High
Uncertainty category: Low
This parameter is the length needed to be achieved before a juvenile matures, from Baker (1998). This parameter acts in a similar, though reverse way to the JUVENILE_DAILYLENGTHGAIN and comes from the same laboratory study; uncertainty of the parameter value is low.



8.6.1.13 LARVA_DAILYGROWTHINCREMENT

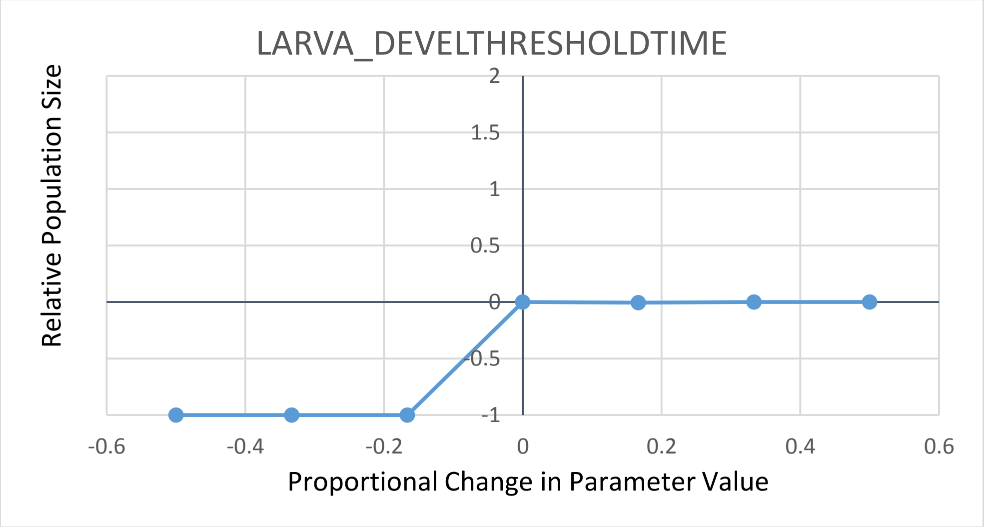
Default value: 0.492

Sensitivity category: Medium
Uncertainty category: Medium
This is the daily increment assuming a linear growth model to the juvenile start length. This is 42.3mm - start length of 1mm divided by 12 weeks. Default value is therefore (42.3 - 1.0) / (12.0*7.0). Although the shape of the curve may not be linear in reality the overall measurement is based on a large data set and is considered reliable. However, the rate of growth is probably dependent upon temperature and food availability in the wild, and this variation is not modelled.



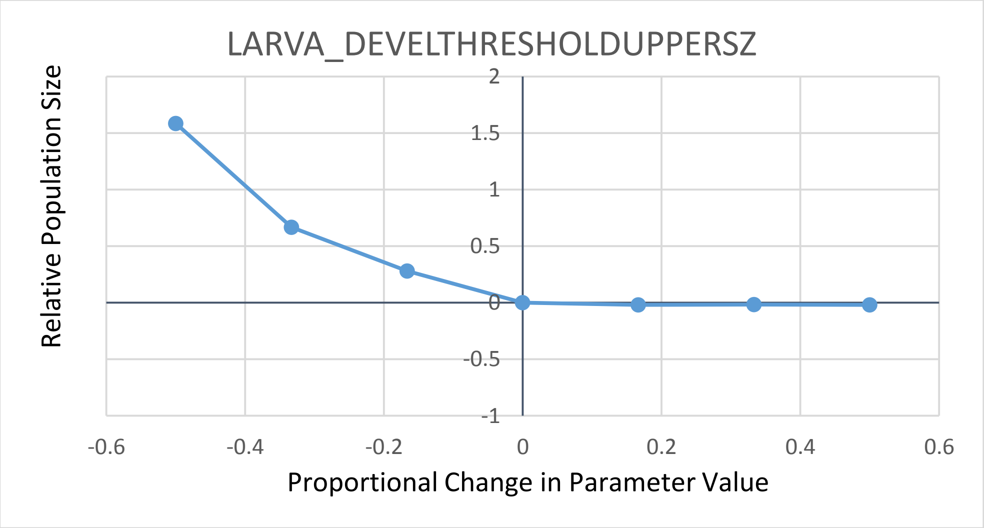
8.6.1.14 LARVA_DEVELTHRESHOLDTIME

Default value: 84
Sensitivity category: Very Low
Uncertainty category: Low
This parameter limits the length of time it is possible for a larva to take to develop. In the current version it is a result of the value used to determine LARVA_DAILYGROWTHINCREMENT, thus if set lower than this the larvae cannot develop. It is therefore a pseudo parameter, not considered further in this version of the model. 84 days (12 weeks) is based on the assumption of 16 weeks total development time on average from (Langton, Beckett et al. 2001).



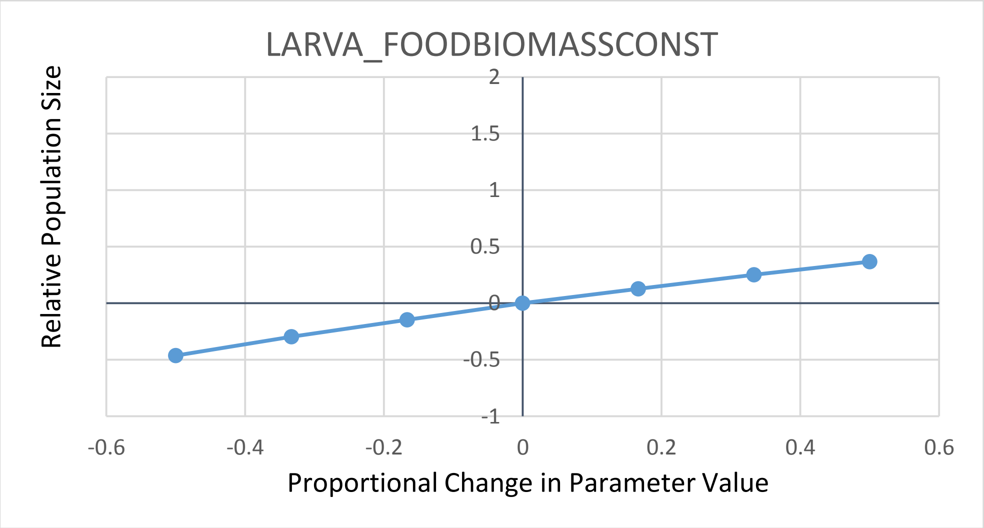
8.6.1.15 LARVA_DEVELTHRESHOLDUPPERSZ

Default value: 42.3 mm
Sensitivity category: Medium
Uncertainty category: Low
This is the upper size at which larvae will undergo metamorphosis. Default value is 42.3 (Baker 1998). The value is considered to be quite accurate based on experimental data.



8.6.1.16 LARVA_FOODBIOMASSCONST

Default value: 215
Sensitivity category: Low
Uncertainty category: High
A constant relating the proportion of food units per m^2, closely linked to LARVA_FOODFOODR. The value is calibrated to estimates of newt density.

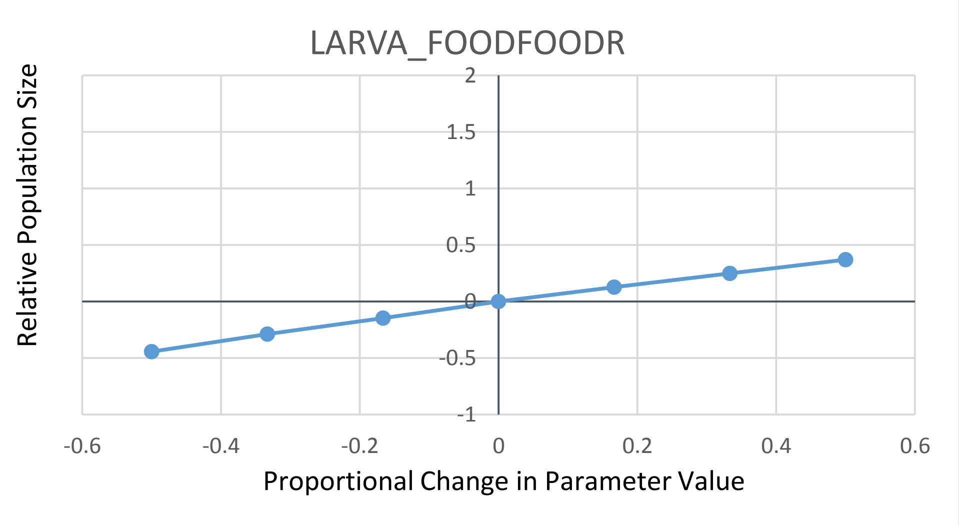


8.6.1.17 LARVA_FOODFOODR

Default value: 0.15
Sensitivity category: Low

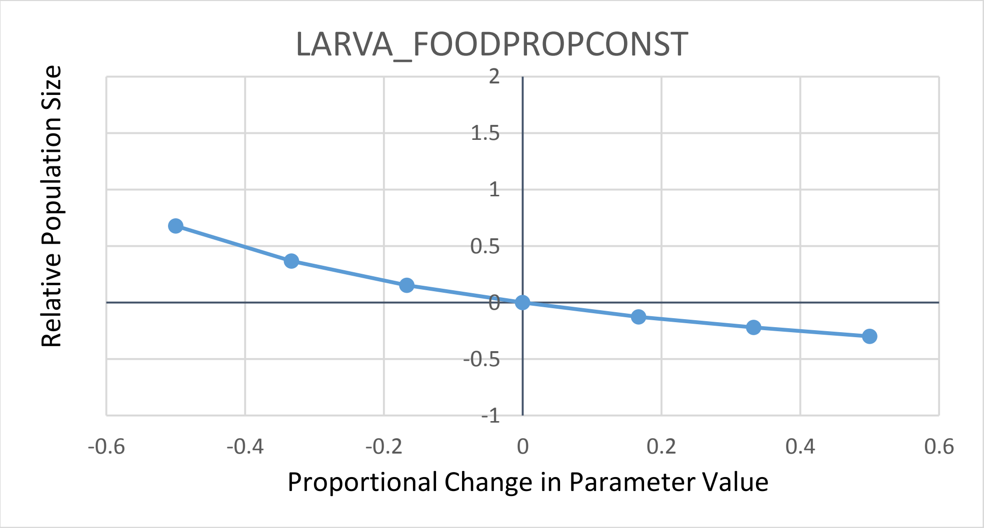
Uncertainty category: High

The instanteous rate of growth for larval food (r from logistic equation), a fitting parameter. This parameter is combined with LARVA_FOODBIOMASSCONST to determine the daily growth rate of larval food. The value is calibrated to estimates of newt density.



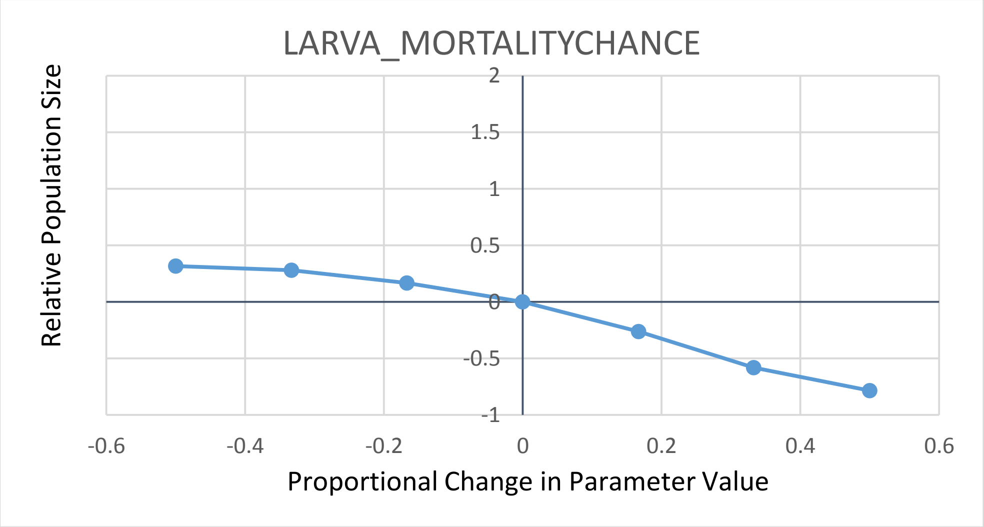
8.6.1.18 LARVA_FOODPROPCONST

Default value: 0.01
Sensitivity category: Low
Uncertainty category: High
A scaling parameter, this is multiplied by larval age is the larval food consumption units of food per day. This is linked to LARVA_FOODBIOMASSCONST and LARVA_FOODFOODR which provide the rate of growth of larval food biomass. The value is highly uncertain, but is calibrated along with the other two parameters based on overall newt density.



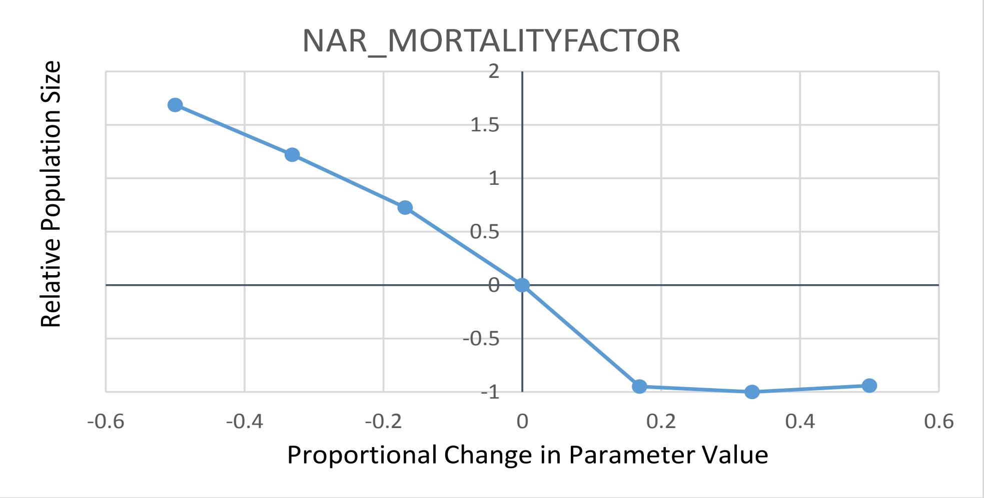
8.6.1.19 LARVA_MORTALITYCHANCE

Default value: 0.005
Sensitivity category: Low
Uncertainty category: Medium
This parameter represents the daily probability of death of a larva not associated with starvation or environmental mortality. Fitted to the assumption of 95% mortality between embryo and metamorphosis (Hedlund 1990). The 95% mortality assumption is therefore based on a single Swedish study, and the extent to which it is general is not known. It does not seem to be considered unrealistic by literature sources, if a little high. Decreasing the values has little effect however.



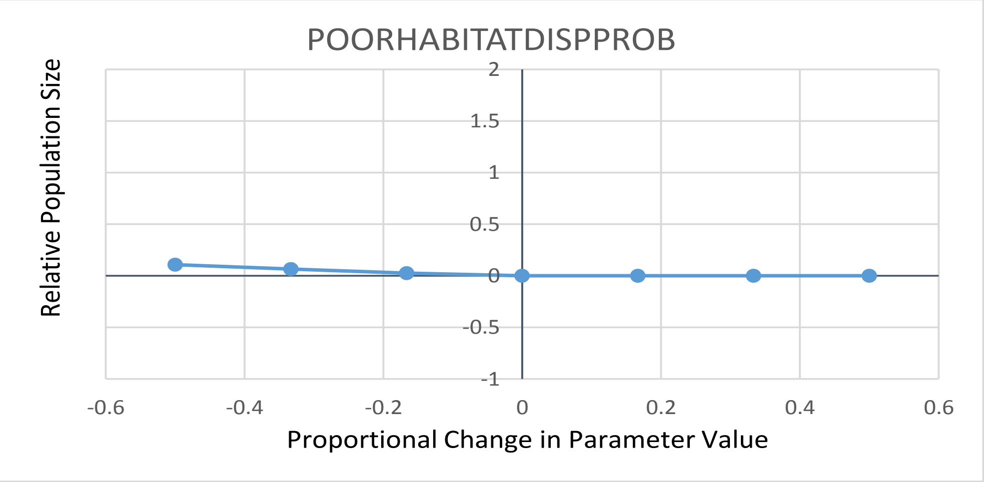
8.6.1.20 NAR_MORTALITYFACTOR

Default value: 0.0016
Sensitivity category: Very High
Uncertainty category: Medium
This is the decrease in survival of overwintering newts per mm non-aquatic period rainfall between January-February and June-December each year from Griffiths, Sewell et al. (2010). The oveall survival chance is ditributed evenly per day over the year following the equation survivalchance = (1.0 - ((nar_rainfall - 200)*NAR_MORTALITYFACTOR)$^{1.0 / 365.0}$, where nar_rainfall is the rainfall in mm, and 200 is given as a lower limit of calculation (rainfall is restricted to the range 200-650mm found by Griffiths, Sewell et al. (2010)). Whilst the numeric value is of low uncertainty, the extent to which the UK study is representative of the wider newt distribution is not known.



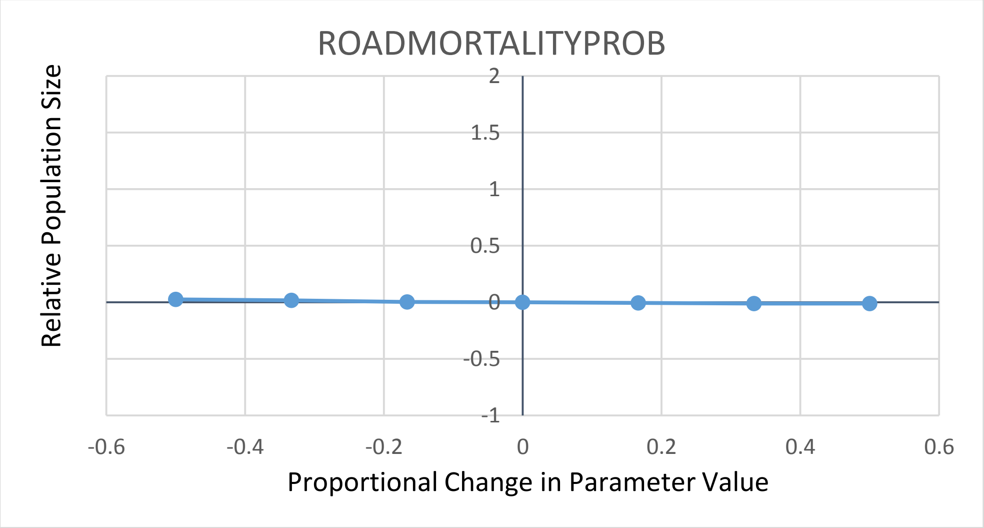
8.6.1.21 POORHABITATDISPPROB

Default value: 1.0
Sensitivity category: Very Low
Uncertainty category: High
This parameter is the probability of free-living newt dispersal if the newt is in poor habitats. Decreasing this chance increases population size slightly, but the parameter is of minor importance. The default value of 1.0 effectively forces movement away from these habitats. Note that decreasing this value increases population size, but this is because there is no negative consequence built into the model of being in a poor habitat (i.e. there is no mortality consequence).



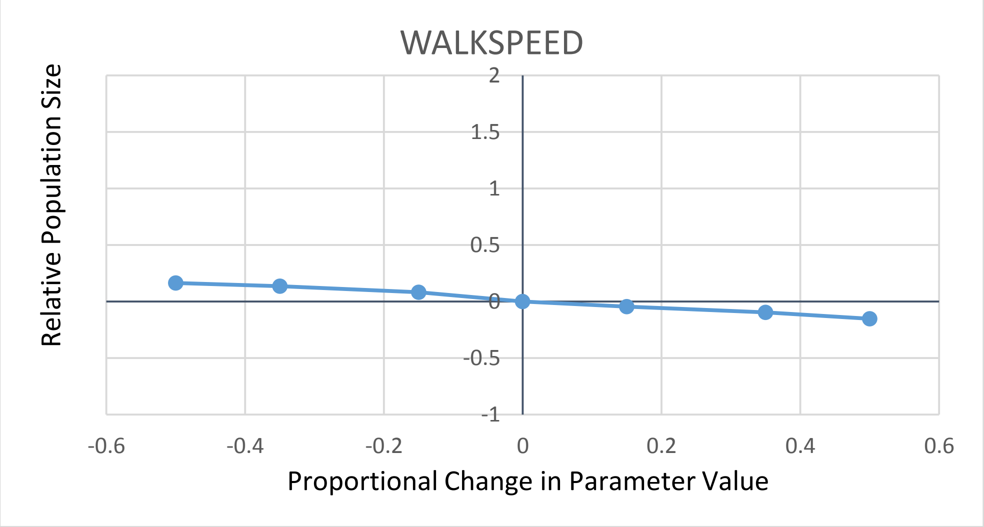
8.6.1.22 ROADMORTALITYPROB

Default value: 0.05
Sensitivity category: Very Low
Uncertainty category: Medium
This parameter is the chance of mortality per road crossing Default value is 0.45 for a typical road, assumed to be 0.9 for busy roads (Hels and Buchwald 2001). The population size appears to be insensitive to this parameter.



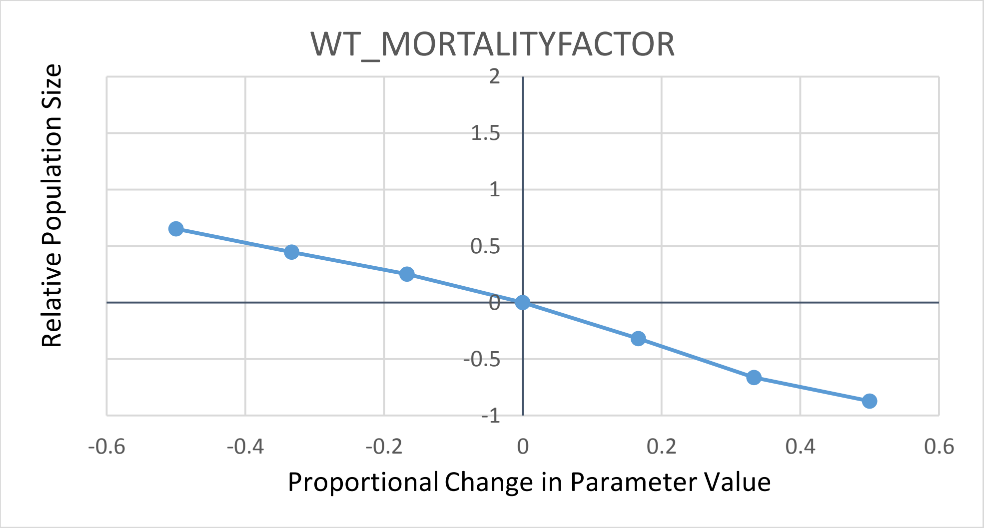
8.6.1.23 WALKSPEED

Default value: 20 m
Sensitivity category: Very Low
Uncertainty category: Medium
This parameter controls the distance moved per day when a newt disperses. The value of 20 comes from Jehle and Arntzen (2000), but other authors also record longer migrations and suggest 1km per year (Arntzen and Wallis 1991, Halley, Oldham et al. 1996). (Kovar, Brabec et al. 2009) estimated spring migration distances at 105-866m. In a radio-telemetry study of *Triturus cristatus*, Jehle (2000) found that more than 50% of adult newts leaving breeding ponds utilised refuges within 15m of the water and that 95 % could be found within 50m of the pond (but note that Jehle & Arntzen recorded examples of daily movement of up to 136.7m). The value of 20 allows for longer juvenile migration and is assumed to be reduced by 90% for adults (following Karlsson, Betzholtz et al. (2007)).



8.6.1.24 WT_MORTALITYFACTOR

Default value: 0.10
Sensitivity category: Medium
Uncertainty category: Medium
This is the per degree decrease in survival in winter minimum temperature (from Griffiths, Sewell et al. (2010)). This value is based on a single, although comprehensive UK study. Different adaptations to local conditions may affect this value in other areas of the newt's distribution.



8.6.2 Parameter uncertainty overview

8.6.2.1 ADULT_LIFESPAN

This parameter has virtually no effect on the simulation since newts are not expected to reach this age. It is effectively redundant due to the value of newt mortality probability parameters. This is as expected for newt survival, which is considered to be low year to year (approximately 50%, (Griffiths, Sewell et al. 2010)).

8.6.2.2 EGG_DEVELDDPARAMETER

This parameter is directly linked to the EGG_DEVELTOTAL. Both parameters show the same degree of sensitivity, but the shape of the mechanism (day-degrees to hatch) is well known and accepted, and the data used to fit the shape of the response curve comes from a similar species, the final values were calibrated to known hatch rates of *T. cristatus*.

Although the model is medium sensitive to these parameters, uncertainty is low and overall, this parameter is considered of minor influence to the model and does not contribute significantly to model uncertainty.

8.6.2.3 EGG_DEVELTOTAL

See above.

8.6.2.4 EGG_MORTALITYCHANCE

Since variation in this parameter is unlikely to be restricted to lower daily values, the overall sensitivity and uncertainty of this parameter is considered to be low due to the low sensitivity of the population size to reduced daily mortality. The reason for the low sensitivity to reduced mortality at the egg stage is density dependent effects at the larval stage, which would in-

crease, almost cancelling out the effect of any increase in egg survivorship.
Overall this parameter is considered of minor influence to the model and does not contribute significantly to model uncertainty.

8.6.2.5 BREEDINGSEASONEND

The effect of varying this parameter is very high, both increase and decrease, hence sensitivity is very high. However, uncertainty is very low for this parameter and large deviations are not expected, so overall uncertainty and therefore model sensitivity are low.

8.6.2.6 DORMANCYTEMP

The uncertainty around this parameter is very low but the model is medium sensitive to its value. Overall it is not considered a very sensitive parameter, except for the unusual response to altering the parameter value either side of the default value. This indicates that there is an interaction between this value and the specific weather pattern used, and development or mortality parameters. Therefore, despite the low population response there is potential for interactions with other parameters and this parameter should be considered important.

8.6.2.7 EGG_DAILYPRODUCTIONVOLUME, BREEDINGSEASONSTART, DORMANCYHUMIDITY, GOODHABITATDISPPROB, ROADMORTALITYPROB, POORHABITATDISPPROB & WALKSPEED

Sensitivity to these parameters is very low, and uncertainty is also medium to low, hence overall, these parameters are considered of no significant influence to the model and do not contribute significantly to model uncertainty.

8.6.2.8 Other parameters

Although individual sensitivity and uncertainty in these parameters varies from very low to very high, there is an overall controlling mechanism that limits the overall population response, meaning that these parameters should be considered as a group for emergent population processes.

Growth parameters for larvae and juveniles all act in a similar way, speeding up or lengthening the time to maturity. Since the overall time to maturity is well documented, and the values are calibrated to this, the overall uncertainty of this period is low, and parameter values certainty is relatively high. Any inaccuracy in one parameter value will be compensated for by others to create a realistic time to maturity. Therefore the following parameters are considered to have an overall low uncertainty as far as emergent population patterns are concerned: JUVENILE_DEVELSIZE, JUVENILEDAILYLENGTHGAIN, LARVALFOODFOODR, LARVA_DAILYGROWTHINCREMENT, LARVALFOODPROPCONST, LARVALFOODBIOMASSCONST, LARVA_DEVELTHRESHOLDTIME. Note however, that because of this use of calibration, the model is not suitable for evaluating changes in developmental times driven by environmental factors.

8.6.2.9 NAR_MORTALITYFACTOR

This parameter has a very high sensitivity and medium uncertainty and having the highest overall uncertainty/sensitivity combination; it is the most critical parameter in the model for further verification. However, like the WT_MORTALITYFACTOR this parameter is a very difficult one to measure requiring a large and long-term study. The value in the study is rather certain, therefore the question is to what extent this study is representative of the newt response over the non-UK part of its range.

8.6.2.10 WT_MORTALITYFACTOR

The parameter is one of two important parameters that determine adult mortality. It is of medium sensitivity and medium uncertainty and therefore is considered to be overall to have a medium to high uncertainty.

8.6.3 Uncertainty Synthesis

As can be seen from the graphs in the sensitivity section, the newt response to changes in parameters are with one exception monotonic and linear in most cases. This indicates that the model is rather stable and predictable. However, parameters are only one part of the model that need to be subjected to uncertainty analysis. This section covers the ways in which parameters are combined, and the uncertainty surrounding the mechanisms built into the model.

8.6.3.1 Development time

There are a large number of parameters related to development, but the actual variation in development time possible is minimal. This is because the model assumes a fixed overall development time based on literature data, and the role of the parameters and mechanisms of development is to recreate a standard phenology. Therefore, the uncertainty in this process is determined by the uncertainty surrounding the overall development time, which is low. However, as noted above, the model is not suitable for use in determining environmentally driven variation in development time or effects associated with altered development time.

8.6.3.2 Mortality

Mortality is the major driver of the model dynamics. There are two key components to this. The first is larval mortality, which is assumed to be density dependent and although the parameters associated with it only show a medium sensitivity, the mechanism of larval mortality is critical to the model, since this is the only density-dependent process included. Egg mortality is highly uncertain, but not thought to be density dependent and is subsumed under the total mortality expected from egg to juvenile stages.

The second important component is mortality of the free-living stages. Weather induced mortality via temperature and rainfall relationships are the most sensitive parameters and mechanisms in the model. Although there are no direct feedback mechanisms which might lead to high model sensitivity, large changes in mortality can interact with density-driven survivorship (e.g. high larval density can result in very low survival, which might be further reduced by weather mortality after emergence). This mortality is combined with a spatial limitation on recolonization, such that if sub-populations become extinct and inter-pond distance is large, the population may be restricted to core areas only.

8.6.3.3 Reproduction

Reproduction is in the model purely a function of numbers of females reaching the pond. As such, any changes in reproductive rates are an indirect result of changes in mortality or distribution of the newts. Since egg production rates are well known, the effect of reproductive process variation in the model is minor. If, however, reproductive rates can be linked to physiological state if the newt in the future, then this process may increase in importance. The use of a constant egg production rate will provide the potential for bias caused by density and timing interactions in larval stages and for early or late arriving breeding newts.

8.6.3.4 Dispersal

Dispersal has very little impact on the population size in terms of sensitivity of the parameters. However, dispersal is essential for situations where spatially distributed catastrophes occur. Therefore, dispersal ability cannot be disregarded and will be of critical importance for evaluation of recovery after local extinction. Dispersal also has an impact on reproduction, since newts that take a long time to find or return to a pond will have a shorter breeding season, and thus may no produce all their eggs. It is not known whether this is true for real life, or simply an artefact of the implementation of reproduction as a constant rate of egg production in the model.

8.6.3.5 Future model development to reduce uncertainty

There are four main areas that need to be addressed in future model development:

1) The most critical for the current implementation is to determine the extent to which weather induced mortality of adults as determined by Griffiths, Sewell et al. (2010) is applicable to the a wider geographical range. If not, it will be necessary to determine how the model can take geographical location into account when determining these parameters.
2) The density-dependent process built into the model is larval food limitation. This appears to be justified as a general rule of thumb, but there are only a few studies that address this (e.g. Walls 1998). In other newt species density-dependence has been shown in the pond for adult stages (Gill 1979), but this has not been recorded for *Triturus* species. Further research into realistic density-dependent mechanisms is needed and further development to justify the larval food limitation mechanism included here would reduce a key uncertainty considerably.
3) Mortality rates of the aquatic stages are calibrated to 95% based on a single Swedish study (Karlsson, Betzholtz et al. 2007), although this fits well with estimation of larval mortality from the smooth newt *Triturus vularis* of 77.3%, however, egg survival in that study was considerably lower giving overall mortality of much more than 95% (Bell and Lawton 1975) . Factors affecting this and other measures from other studies would increase realism of the mortality processes.
4) It would be extremely useful to know to what extent a standard development time is justified. Assuming this is plastic, then to improve the usefulness of the model, enabling a mechanistic developmental rate changes it is necessary. Assuming this is not fixed, then to determine how to include mechanistic variation in the model. In doing this the most critical component appeared to be processes affecting juvenile development in the terrestrial phase.

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Appendix 1. Life-history strategy ERA modelling attempt

General animal model construction was completed as originally planned for the project. Documentation of the model development and code can be found here:
https://almassdocs.au.dk/ALMaSSODdox/MixNMatchODdox/index.html
The strategy chosen was to create generalised life stages (e.g. egg, larva, mobile adult) and provide these with standard behaviours relying on externally generated parameters. A species model was then built by combining a set of life-stages and the respective values for its parameter values for each stage. The original concept of the project was that these parameter values could be varied within realistic ranges and thus a full range of life-history strategies could be tested. The code was developed to create these definable species (dubbed mix-N-match animals). The code was then tested and debugged before attempting to create a number of life-histories representing 'generic species' of interest for pesticide ERA testing.
To create a mix-n-match species it is necessary to first determine how many life-stages are needed (e.g. egg, juvenile, adult) and then to parameterise the standard set of parameters for each of those life-stages.
The code works from the point of view of code functionality and it was possible to create simple models representing e.g. skylarks, beetles and spiders, but not possible to make suitable generic representation of other life-history strategies of interest e.g. newts with complicated migrations linked to life-stages. The full C++ code is available from
http://ccpforge.cse.rl.ac.uk/gf/project/almass/scmsvn/

Figure 1 shows an example of a mix-n-match species where habitat specificity for grassland types restricts its spatial distribution. Figure 2 shows an example of a mix-n-match 'bird' output when we assume density dependence only acts on adults causing them to disperse to find breeding (effectively mimicking territoriality).

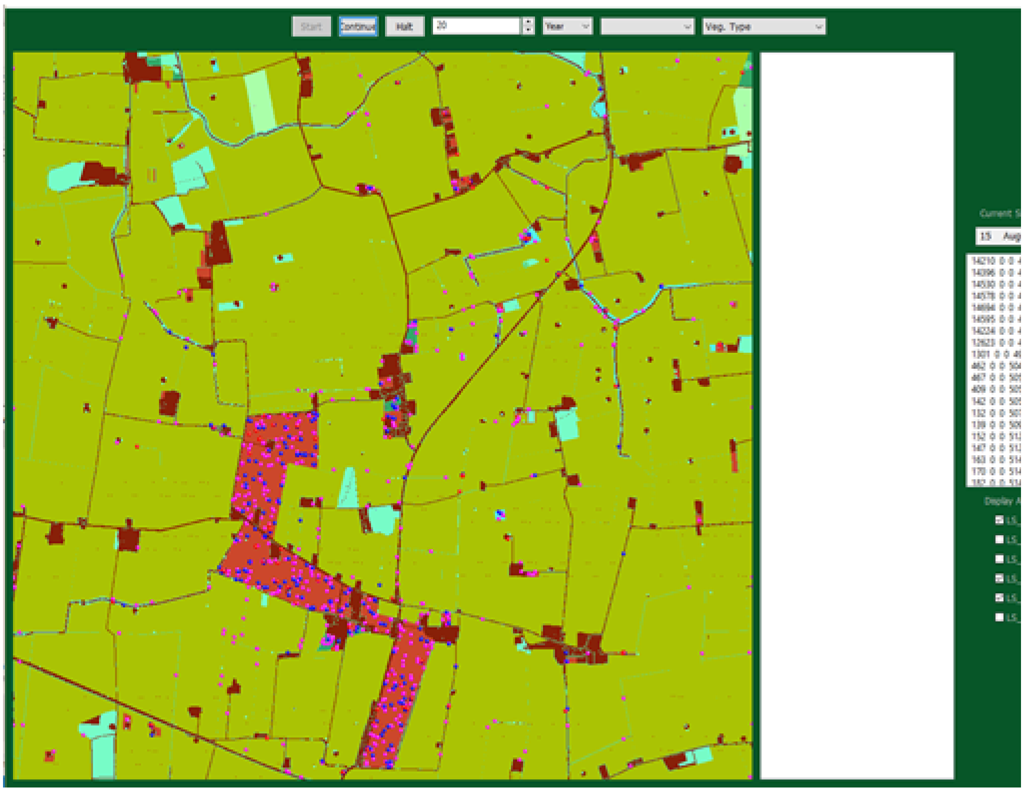

Figure 1. Example ALMaSS run showing a mix-n-match animal (pink & blue dots) that is primarily associated with grassland and grassy boundaries and areas.

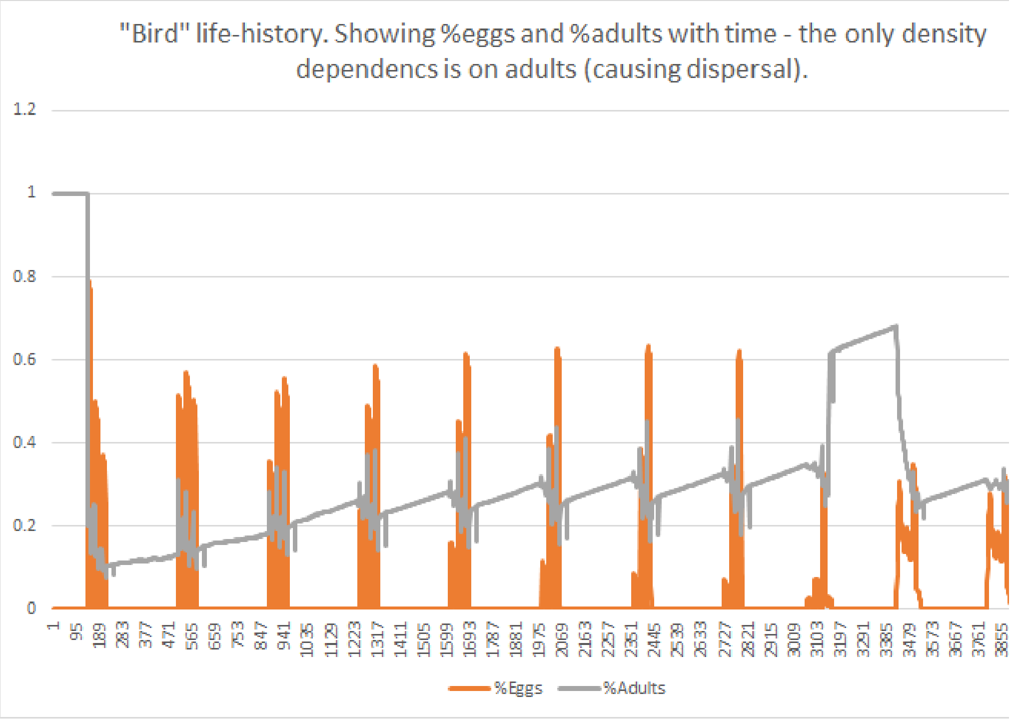

Figure 2. Example population structure output of a 'bird' mix-n-match species (x-axis is simulation days, y-axis is proportion of the population).

Appendix 1.1 Problems arising

During the testing and model development procedure a number of issues became apparent:

- The generic models although simple had rather complicated parameter sets to define them because they needed to take account of a very large degree of variability.

This results in the need for the following parameters defined for each and every life-stage (many will be set to zero, i.e. not used):

  - LifespanDays
  - LifespanDayDegrees
  - MortalityChance
  - MovementDailyMax
  - MovementDailyMin
  - MovementDispersalMax
  - MovementDispersalMin
  - ReproductionEndDate
  - ReproductionStartDate
  - ReproductionLitterSize
  - ReproductionFrequency
  - ReproductionAnnualOutput
  - PesticideThresholdConcentration
  - PesticideMortalityRate
  - PesticideReproductionReductionChance
  - PesticideReproductionEffectType
  - PesticideReproductionReductionMagnitude
  - CompetitionType
  - CompetitionParameter1
  - CompetitionParameter2
  - CompetitionParameter3
  - For each habitat type in ALMaSS the affinity for that habitat is defined for each life-stage on a scale of 0-4 (4 = optimal).
- It was not possible to create realistic models from these parameters (they are supposed to be general). However, this means that the life-history types identified in Task 2 as being important could not all be modelled. For example newt migration to and from ponds, pollinator attraction of flowering fields and bat foraging behaviour would not be possible to simulate using these models. However, in all three cases these behaviours are critical to evaluation of the threat posed by pesticide use.

- The original project idea was to evaluate a full range of parameter values for each parameter in the life-history, creating a massive matrix of results for regression tree analysis. However, it is clear that this presents a number of practical problems. The first and most obvious of these is that real species do not have the range of possible parameters, but a subset of these. The subset is not random but linked to ecological and physiological traits e.g. large animals long-lived animals cannot produce large numbers of offspring. This means that evaluating which subsets of parameters to test would be a large and difficult job.

- The simple models appeared to be rather robust in terms of population elasticity. Details of the ecology and behaviour which may cause them to be susceptible to pesticide effects were not possible to include e.g. the pollinator attraction to crops, or detail of phenology. In addition, the combined species model often gave unrealistic population dynamics e.g. drifting of breeding season due to reliance on development days. Figure 2 shows an example of this, where the ‘bird’ population builds up steadily then crashes. This is clearly neither realistic nor desirable, but this type of effect very difficult to predict. This is a function of the individual based format to the models and the need to specify developmental feedbacks explicitly contra population models where feedbacks are subsumed under density-dependent relationships on mortality and reproduction, and development is usually simply assumed as an annual time-step.

- During the project, considerable time and effort was used to generate detailed maps of Danish landscapes as well as the farming regimes contained therein. These maps could not be used by generalised models e.g. the mix-n-match bird could not respond to management or crop height, however, we know that these factors are critical to some birds (e.g. skylarks). These factors arise from the map and the simulation of farming, and therefore much of the effort used to generate realistic drivers for landscape context was not utilised when using the mix-n-match model approach.

Taking these issues as a whole suggests that population risk assessment is be definition a species based approach, and that what makes a species a species is necessary to include in the models.

The consequence of this was that the Mix-n-match approach was abandoned for risk assessment in this project and a series of species specific case studies were created to highlight new issues resulting from the policy change in regulatory risk assessment from individual toxicity to populations in a landscape context.

Developing spatio-temporal models for landscape-scale pesticide ERA

This report investigates various aspects of landscape-scale population-level environmental risk assessment (ERA) for the purposes of regulatory risk assessment for pesticides at EU and member state level.
We provide examples of landscape-scale population-level ERA for birds and mammals, and terrestrial invertebrates using existing models for the European Brown Hare (Lepus europeaus), the Skylark (Alauda arvensis), and a carabid beetle (Bembidion lampros). We also develop a model for Great Crested Newt (Triturus cristatus). Amphibians are not currently addressed in regulatory ERA but are assumed to be covered by bird and mammal risk assessment. This situation is now under review by EFSA, therefore the newt is a likely candidate for landscape-level risk assessment in the future.
The main result of the simulations confirms that landscape context has a very large influence on the results of an ERA. In addition, the results were not transferable between species, and species ecology interacted with landscape context to create reproducible but unpredictable variation in ERA under different conditions. As a consequence, the concept typically used in ERA of realistic worst-case scenario cannot be generally applied, but must be created for each specific set of contexts.

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