Linking the presence of invasive alien species to measures of ecological quality

Chief Scientist’s Group report

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Professor Doug Wilson
Chief Scientist

Executive summary

The Environment Agency is responsible for implementing the Water Framework Directive (WFD) in England, including establishing river basin management plans to manage water bodies that are failing to achieve the target of good ecological status. Management plans have to consider invasive non-native species (INNS) where they are causing sites to fail to achieve good ecological status. As measures are put in place to reduce pressures on water bodies other than those caused by INNS, it is likely that that any underlying impacts of INNS will become more visible. This raises 2 important issues for the Environment Agency, namely:

- What impacts do INNS have on the ecological health of sites at which they are recorded?
- Do the tools used to measure ecological status for the WFD reflect the impacts of INNS?

About the project

The aim of this project was to use the available evidence to better understand the impacts of INNS on the ecological status of water bodies as measured by the WFD tools. The project addressed the following key questions.

- Is any effect of INNS reflected in measures of the ecological status of a water body as measured by the WFD tools?
- Which WFD tools are likely to respond to each particular INNS species?
- Is it possible to identify when these biological tools may have ‘missed’ an impact or provided a false signal?

The measure of ecological quality used to classify the ecological status of sites according to the WFD is the Ecological Quality Ratio (EQR). Any significant impact of INNS on the EQR will influence the ecological status of the water body, with a consequent impact on WFD objectives. As INNS have the potential to cause changes that propagate through ecosystems affecting multiple components of the community, it was important to ensure that data on all biological quality elements (macroinvertebrates, macrophytes, fish) were analysed where possible.

Statistical analyses were applied to Environment Agency data gathered during operational monitoring from river, lake and canal water bodies throughout England for:

- macrophytes – derived by the WFD LEAFPACS2 tool
- fish – derived by the Fisheries Classification Scheme 2 (FCS2) tool
- macroinvertebrates – derived by the River Invertebrate Classification Tool (RICT)

A list of relevant INNS was compiled from assessments by the WFD United Kingdom Technical Advisory Group (WFD UKTAG) and the Joint Nature Conservation Committee (JNCC) and their presence in water bodies determined from Environment Agency data and data obtained from the National Biodiversity Network.

To assess the extent to which INNS have become established in water bodies across England, the number of INNS recorded between 2009 and 2017 in each river, lake and canal water body was determined. The distribution of higher numbers of INNS was found to coincide with regions where other stressors are often high, suggesting that care is needed to separate the impact of INNS from those of other stressors.

To illustrate the current distribution of INNS relevant to the Environment Agency, data on species occurrences matched to water bodies were used to determine the area of extent for each species based on the approach used to develop the England Vascular Plant Red List.

Attributing any difference in measures of ecological quality to INNS through later data analysis is difficult. To increase the probability of detecting differences in measures of ecological quality and attributing any impact to invasive species, analysis was conducted at 2 different scales.

- Reach scale over individual years. This provided high confidence that the INNS was present/absent at the site at the time of sampling, but less confidence when attributing causality to the presence of INNS, such that statistical tests were able to identify differences that were associated with the presence of the INNS.
- Water body scale over WFD reporting periods. The confidence that the INNS was present/absent at the site at the time of sampling was lower, but there was greater confidence when attributing causality to the presence of INNS through asymmetric analysis of variance following a before–after–control–impact (BACI) design with multiple water bodies within each of the impacted and control groups.

In both cases, EQR data from 2003 to 2014 from sites with the INNS present were compared with similar sites where the INNS were absent.

Key findings and their implications

There was strong evidence that 2 of the species tested (signal crayfish, *Pacifasticus leniusculus*, and demon shrimp, *Dikerogammarus haemobaphes*) have substantial impacts on the WFD measures of ecological quality. These species were found to have resulted in an effective reduction of EQR equivalent to approximately half to three-quarters of a WFD class. It is likely that other INNS have an impact on WFD measures of ecological quality, although the confidence in the evidence is less strong. All the INNS tested except common carp (*Cyprinus carpio)* showed some evidence of a difference in measures of ecological quality where they were present.

The impact of INNS becomes more pronounced with the length of time that the species has been present. This finding has operational implications since the WFD tools may not detect any impact of INNS for some time after initial invasion. By this time the INNS is likely to have established a substantial population and be harder to deal with.

Understanding the mechanism by which the INNS causes an impact on measures of ecological quality is confounded as many INNS are included in the list of taxa used to measure ecological quality. All the WFD tools investigated were affected. Hence the occurrence of an INNS may have a positive or negative arithmetic influence on the measure of ecological quality, depending on how they are perceived within the tool, with the effect not based on a real biological impact on the quality of the site. Such influences of INNS on measures of ecological quality will have operational implications, as the occurrence of INNS is likely to confound interpretation of other stressors, potentially leading to inappropriate programmes of measures.

It is therefore suggested that further analyses are made where EQR is calculated by excluding INNS to provide a cleaner signal of the impact of the species on the ecological quality of the site. Community level analyses should also determine what impacts INNS have at the community/species level.

The current WFD UKTAG system for classifying surface water bodies based on the presence of high impact alien species may lead to ‘double accounting’ for INNS where impacts are already apparent or a plus/minus effect where the tools are confounded.

Acknowledgements

The project team is grateful to the Botanical Society of Britain and Ireland for providing access to its data and in particular to Dr Kevin Walker, Head of Science, for facilitating this.

The team is also grateful to the GB Non-Native Species Secretariat, particularly Olaf Booy, for providing the modified EICAT grades of potential risk presented by INNS in the UK.

Finally, the project team would like to say a considerable thank you to all the Environment Agency staff involved in the project, in particular Dave Smith, Jon Barrett and the members of the Invasive Species Action Group.

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1 Introduction

1.1 Background to the project

Human activities are the main driver responsible for the transfer at an unprecedented rate of species among regions (Seebens et al. 2017). As invasive non-native species (INNS) often constitute new functional components of communities, they can cause enormous change in recipient ecosystems (Ricciardi et al. 2013, Gallardo et al. 2016) and are a leading cause of native animal extinctions (Clavero and García-Berthou 2005). Impacts of INNS are not restricted to those on native, functionally equivalent or prey species, rather they have the potential to cause changes that propagate through ecosystems affecting multiple components (see, for example, Roy et al. 2014, Kratina and Winder 2015, Roy and Brown 2015; see Figure 1.1).

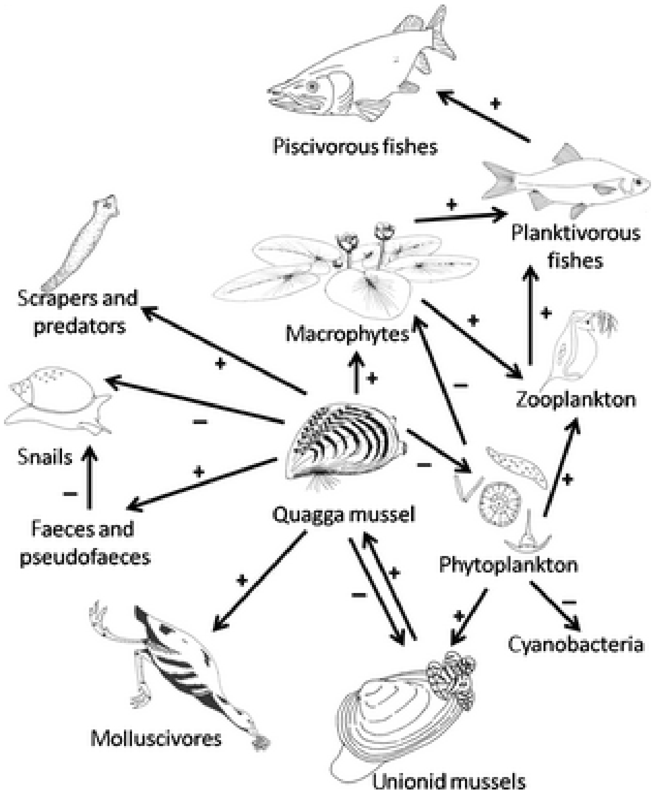

Figure 1.1 Possible direct and indirect effects of INNS on Britain's freshwater ecosystems using quagga mussels as an example

Source: Roy et al. (2014)

Identification of the impacts caused by INNS is a critical step in risk assessment and subsequent prioritisation for management. The importance of identifying the impacts of INNS is highlighted by:

- Convention on Biological Diversity's Aichi Target 9[1]

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[1] 'By 2020, invasive alien species and pathways are identified and prioritised, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment' (www.cbd.int/sp/targets/rationale/target-9/).

- EU Biodiversity Strategy Target 5 – Combat alien species[2]
- EU Regulation 1143/2014 on INNS (the IAS Regulation)[3]

These requirements are acknowledged within the GB Non-native Species Strategy (GBNNSS 2015). The GB Non-Native Species Information Portal[4] funded by Defra supports the strategy through documenting impacts. However, strong empirical evidence of impacts is often lacking and so there can be a high degree of uncertainty in assessments of impact (Roy et al. 2014, 2018).

The Environment Agency is responsible for implementing the Water Framework Directive (WFD) in England, including drawing up river basin management plans to deal with water bodies that are failing to achieve the target of good ecological status. Management plans have to consider INNS where they are causing sites to fail to achieve good ecological status. As measures are put in place to reduce pressures on water bodies other than those caused by INNS, it is likely that any underlying impacts of INNS will become more visible. This raises 2 important issues for the Environment Agency, namely:

- What impacts do INNS have on the ecological health of sites at which they are recorded?
- Do the tools used to measure ecological status for the WFD reflect the impacts of INNS?

1.2 Project aim and objective

The aim of this project was to use the evidence available to establish the impacts of INNS on ecology and on the ecological status of water bodies as measured by WFD tools such as LEAFPACS2, the Fisheries Classification Scheme 2 (FCS2) and the River Invertebrate Classification Tool (RICT).

The objective was to apply statistical analyses to data that compare the recorded presence of INNS with WFD measures of ecological health on the site at which they are recorded, addressing the following key questions.

- Is any effect of INNS reflected in measures of the ecological status of a water body as measured by the WFD tools?
- Which WFD tools are likely to respond to each particular INNS species?
- Is it possible to identify when these biological tools may have 'missed' an impact or provided a false signal?

To do this, the project considered data collected from river, lake and canal water bodies throughout England. Records of the occurrence of INNS from 1970 to 2017 were used, together with data on ecological status from 2003 to 2014.

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[2] 'By 2020, invasive alien species and their pathways are identified and prioritised, priority species are controlled or eradicated, and pathways managed to prevent new invasive species from disrupting European biodiversity' (http://ec.europa.eu/environment/nature/biodiversity/strategy/index_en.htm).

[3] http://ec.europa.eu/environment/nature/invasivealien/index_en.htm

[4] www.nonnativespecies.org/index.cfm?pageid=408

1.3 Structure of the report

Section 2 explains how the list of INNS considered in the project was compiled, which data sources were used and how the data were screened.

Section 3 describes the approach used to assess the extent to which INNS have established in water bodies across England and how the area of extent of individual species was estimated.

Section 4 presents the methods used to determine if the presence of INNS has a significant impact on ecological quality as measured by the WFD tools and the results of the data analysis. It includes a discussion of the findings and the limitations of the approach adopted.

Section 5 sets out the conclusions from the study and their implications.

Section 6 suggests various areas of further work to obtain a fuller understanding of the impact of INNS on WFD measures of ecological quality.

2 Data compilation

2.1 INNS to be considered

A list of INNS to be considered in this project was compiled using lists of INNS considered to have potential impacts on freshwater aquatic systems given in 2 documents:

- ‘Revised Classification of Aquatic Alien Species according to their Level of Impact’ produced by the WFD United Kingdom Technical Asvisory Group (WFD UKTAG), which lists all INNS with the potential to have an impact on rivers or lakes (WFD UKTAG 2015)
- ‘UK Biodiversity Indicators B6: Pressure from invasive species – technical background document’, published by the Joint Nature Conservation Committee (JNCC), which lists all INNS identified as being freshwater (Harrower et al. 2017)

This exercise produced a list of taxa for consideration (Table 2.1) made up of:

- 41 plant species (25 aquatic macrophytes and 16 riparian species)
- 31 macroinvertebrate species
- 13 fish species

To prioritise these taxa for data screening, they were ranked according to the grades of concern for species in Great Britain allocated to them by the GB Non-Native Species Secretariat (GBNNSS). The system used by the GBNNSS for grading the potential risk presented by INNS is a modified Environmental Impact Classification for Alien Taxa (EICAT) (Blackburn et al. 2014, Hawkins et al. 2015). This provided 5 grades of concern for species where sufficient data were available.

- **Massive (MV).** A species is considered to have massive impacts when it leads to the replacement and local extinction of native species, and produces irreversible changes in the structure of communities and the abiotic or biotic composition of ecosystems. Note that ‘local’ refers to the typical spatial extent over which the original native communities can be characterised.
- **Major (MR).** A species is considered to have major impacts when it causes the local or population extinction of at least one native species, leads to reversible changes in the structure of communities and the abiotic or biotic composition of ecosystems, and has no impacts that cause it to be classified in the MV impact category.
- **Moderate (MO).** A species is considered to have moderate impacts when it causes declines in the population size of native species, but no changes to the structure of communities or to the abiotic or biotic composition of ecosystems, and has no impacts that would cause it to be classified in a higher impact category.
- **Minor (MN).** A species is considered to have minor impacts when it causes reductions in the fitness of individuals in the native biota, but no declines in native population sizes, and has no impacts that would cause it to be classified in a higher impact category.

- **Minimal Concern (MC).** A species is considered to have impacts of minimal concern when it is unlikely to have caused deleterious impacts on the native biota or abiotic environment. Note that all alien taxa have impacts on the recipient environment at some level such as by altering species diversity or community similarity (for example, biotic homogenisation), and for this reason there is no category equating to ‘no impact’.

Species with insufficient data to classify them were graded **Unclassified (UC)**.

All invasive species to be considered were allocated the grades established by the GBNNSS (Table 2.1). As the grading was merely a mechanism for focusing effort during data screening, the confidence in these scores was not used. Details of the modified EICAT grading system used by the GBNNSS, including details of the decision-making process and the working groups used to derive the grades are given in Roy and Booy (2016).

The EICAT system for grading the potential risk presented by INNS is iterative and is dependent on the evidence available at the time of grading. As such, the outputs of this project should inform future grades.

Table 2.1 INNS considered in this project with their GBNNSS-modified EICAT grade of concern

Macrophytes		Riparian plants		Invertebrates		Fish	
Cabomba caroliniana	**MR**	***Rhododendron ponticum***	**MV**	***Corbicula fluminea***	**MV**	***Cyprinus carpio***	**MR**
Crassula helmsii	**MR**	***Claytonia sibirica***	**MR**	***Dreissena bugensis***	**MV**	***Pseudorasbora parva***	**MR**
Hydrocotyle ranunculoides	**MR**	***Fallopia japonica***	**MR**	***Dreissena polymorpha***	**MV**	*Ameiurus melas*	MO
Lagarosiphon major	**MR**	***Fallopia x bohemica***	**MR**	***Astacus leptodactylus***	**MR**	*Carassius auratus*	MO
Ludwigia grandiflora	**MR**	*Crocosmia pottsii x aurea*	MO	***Cordylophora caspia***	**MR**	*Lepomis gibbosus*	MO
Ludwigia peploides	**MR**	*Fallopia sachalinensis*	MO	***Dikerogammarus villosus***	**MR**	*Leuciscus idus*	MO
Lysichiton americanus	**MR**	*Heracleum mantegazzianum*	MO	***Eriocheir sinensis***	**MR**	*Salvelinus fontinalis*	MO
Mimulus guttatus	**MR**	*Impatiens glandulifera*	MO	***Gammarus tigrinus***	**MR**	*Sander lucioperca*	MO
Mimulus guttatus x luteus	**MR**	*Lupinus nootkatensis*	MO	***Hemimysis anomala***	**MR**	*Silurus glanis*	MO
Myriophyllum aquaticum	**MR**	*Petasites albus*	MO	***Orconectes limosus***	**MR**	*Ctenopharyngodon idella*	MN
Aponogeton distachyos	MO	*Petasites fragrans*	MO	***Orconectes virilis***	**MR**	*Leucaspius delineatus*	MN
Egeria densa	MO	*Petasites japonicus*	MO	***Pacifastacus leniusculus***	**MR**	*Oncorhynchus mykiss*	MN
Elodea nuttallii	MO	*Crocosmia paniculata*	MC	***Procambarus clarkii***	**MR**	*Rhodeus sericeus*	MN
Lemna minuta	MO	*Impatiens capensis*	MC	***Rangia cuneata***	**MR**		
Sagittaria latifolia	MO	*Impatiens parviflora*	MC	*Dikerogammarus haemobaphes*	MO		
Eichhornia crassipes	MN	*Rhododendron luteum*	MC	*Potamopyrgus antipodarum*	MO		
Elodea canadensis	MN			*Crangonyx pseudogracilis*	MN		
Acorus calamus	MC			*Astacus astacus*	MC		
Azolla filiculoides	MC			*Branchiura sowerbyi*	MC		
Elodea callitrichoides	MC			*Caecidotea communis*	MC		
Mimulus luteus	MC			*Chelicorophium curvispinum*	MC		
Mimulus moschatus	MC			*Ferrissia (Petancyclus) wautieri/clessiniana*	MC		
Juncus ensifolius	UC			*Girardia tigrina / Dugesia tigrina*	MC		
Mimulus ringens	UC			*Menetus (Dilatata) dilatatus*	MC		
Myriophyllum heterophyllum	UC			*Musculium transversum*	MC		
				Mytilopsis leucophaeata	MC		
				Physella acuta	MC		
				Physella gyrina	MC		
				Planaria torva	MC		
				Hypania invalida	UC		
				Marstoniopsis insubrica	UC		

Notes: Species classified as MR are shown in black and bold. Species classified as MO or MN are shown in black. Species classified as MC or UC are shown in grey.

2.2 Data sources

Data gathered during operational monitoring were supplied by the Environment Agency. Table 2.2 lists the type of data used.

Table 2.2 Type of data provided from Environment Agency operational monitoring

Source	Type of data
Macrophyte sampling	• Sample site location • Water body ID • Sample date • Species identified • EQR derived using the WFD LEAFPACS2 tool • Environmental information associated with the sample Data were available on community composition from 1997 to 2017, and EQR from 2004 to 2014.
Fish sampling	• Sample site location • Water body ID • Sample date • Species identified • EQR derived by the FCS2 tool • Environmental information (generally derived from GIS) associated with the sample Data were available on community composition from 1997 to 2017, and EQR from 2003 to 2014.
Macroinvertebrate sampling	• Sample site location • Water body ID • Sample date • Species identified • EQRs (for NTAXA and ASPT) derived by the RICT tool • Environmental information associated with the sample Data were available on community composition from 1997 to 2017, and EQR from 2006 to 2014.
River Habitat Surveys	• Site location • Survey date • Categorical abundance of the riparian invasive species: *Fallopia japonica* (and related species) (Japanese knotwood); *Impatiens glandulifera* (Himalayan balsam) and *Heracleum mantegazzianum* (giant hogweed) Data were available on community composition from 1994 to 2014.
Water body information	• Water body ID (WFD cycle 1 and 2) • Water body EQR • Pressure data

Notes: ASPT = average score per taxon; GIS = geographical information system; NTAXA = number of scoring taxa

In addition, data were obtained from the National Biodiversity Network describing the location and date of records of relevant INNS. Although records were available from earlier, the majority of the data were from 1970 to 2017.

All data were compiled in a bespoke Microsoft SQL server relational database with complete referential integrity, thus ensuring that each record of a species or environmental parameter relates to a single location and occasion.

2.3 Data screening

To determine sites where INNS were present, data were matched spatially to rivers, lakes and canals in ArcMap 10.2. Sample site locations were plotted together with the river, canal and lake networks based on the Ordnance Survey (OS) 1:25,000 map.

The ‘blue line’ of the river network was divided into segments (river reaches) where each segment consisted of a continuous section of a single river channel without any joining channels; divisions between segments occurred at the confluence of channels (tributaries, anastomosing channels). Hence, each segment represented a continuous river channel without inflows (as indicated on the OS 1:25,000 map). Similarly, the canal network was divided into segments where each segment represented a canal pound. Lakes were not subdivided.

Sampling sites were matched if they both intersected with the same river/canal segment or lake. A buffer of 50m was added to the networks (that is, sites were regarded as intersecting with a segment if they occurred within 50m of that segment of the blue line). This allowed for imperfect location of sites and for the fact that the ‘blue line’ may not accurately represent the full width of the channel. However, as sites could be erroneously clipped to a river/canal segment if they were located near a confluence, manual checks were made to avoid such errors.

Due to variation in the spatial resolution of the records held by the National Biodiversity Network, only those data that could be matched to relevant water bodies (that is, rivers, lakes and canals) were included (that is, those data resolved to within 100m^2). Less spatially resolved records (10km^2 to 1km^2) could encompass multiple water bodies, as well as other non-relevant habitats (ponds, ditches, and for riparian species terrestrial environments) and therefore could not be matched to water bodies.

Site identities were then used to match biological data from all sources and water body data using the Microsoft SQL server database.

3 Current distribution of INNS

3.1 Number of INNS per water body

To assess the extent to which INNS have established in water bodies across England (and thus are a potential issue), the number of INNS recorded from 2009 to 2017 in each river (Figures 3.1 and 3.2), lake and canal (Figure 3.3) water body was determined. The number of species within each GBNNSS-modified EICAT grade of concern in each river water body was also determined (Figures 3.4 to 3.9; species graded UC not shown).

The maximum number of INNS recorded in a river water body was 25 (Figure 3.1). River water bodies with higher numbers of INNS tended to be lowland large rivers (for example, the Thames, Severn, Trent, rivers of the Wash) and/or in areas of high population (although the relationship with population size was not tested).

The mode for river water bodies was one INNS per water body (Figure. 3.2a), whereas for lakes and canals the mode was zero INNS per water body (Figure. 3.3). However, lakes and canals have been subject to less sampling effort, reducing the probability of detecting INNS.

The river water bodies that did not contain INNS tended to be headwaters, upland areas (for example, Exmoor, Dartmoor, North Pennines) or coastal river catchments. The mode for coastal river catchments was zero (Figure. 3.2b), with the majority of coastal river catchments falling into this class. Although coastal catchments are less connected than other rivers, these water bodies have been subject to less sampling effort, reducing the probability of detecting INNS.

The distribution of water bodies containing higher numbers of INNS gives some indication of the routes of invasion, suggesting that ports and centres of population have acted as points of entry, with further invasions via movement of INNS through river catchments and between catchments via the canal network. The distribution of higher numbers of INNS coincides with regions where other stressors are often high, suggesting that care will be needed to separate the impact of INNS from those of other stressors.

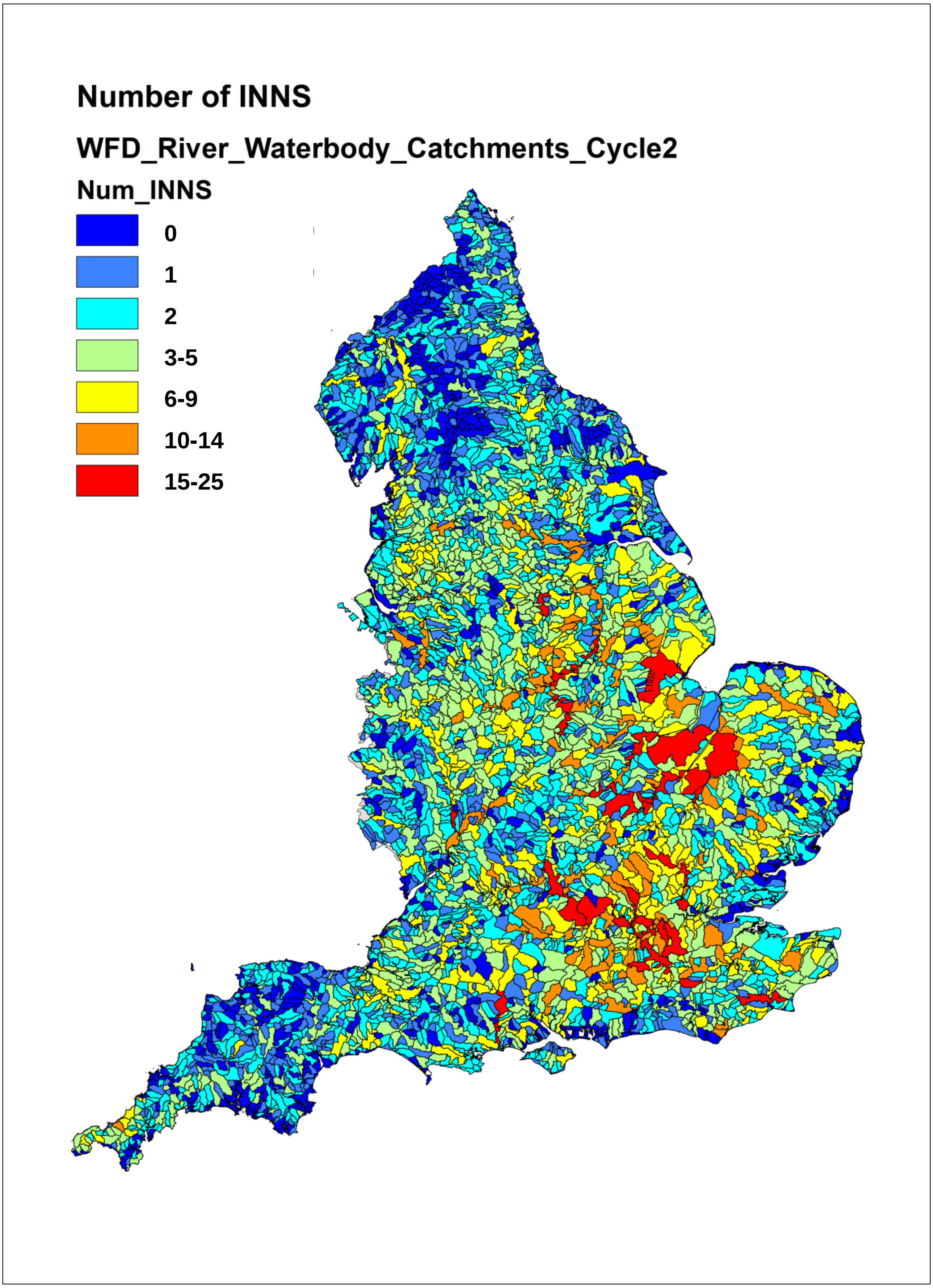


Figure 3.1 Total number of INNS recorded 2009 to 2017 in each river water body catchment (WFD cycle 2)

Notes: All EICAT grades; see Table 2.1 for list of species.

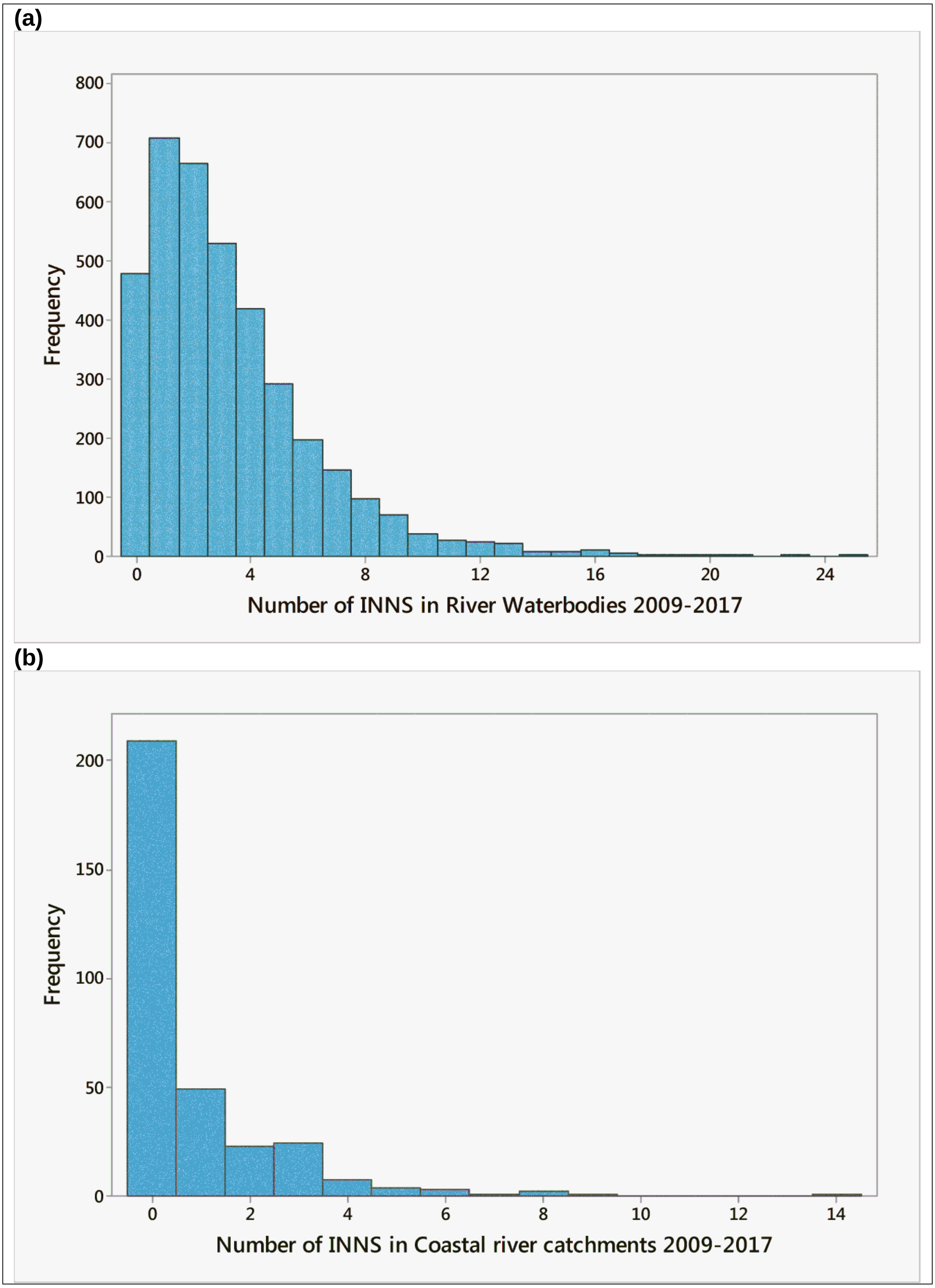


Figure 3.2 Frequency of INNS recorded 2009 to 2017 in (a) all river water body catchments and (b) coastal river catchments

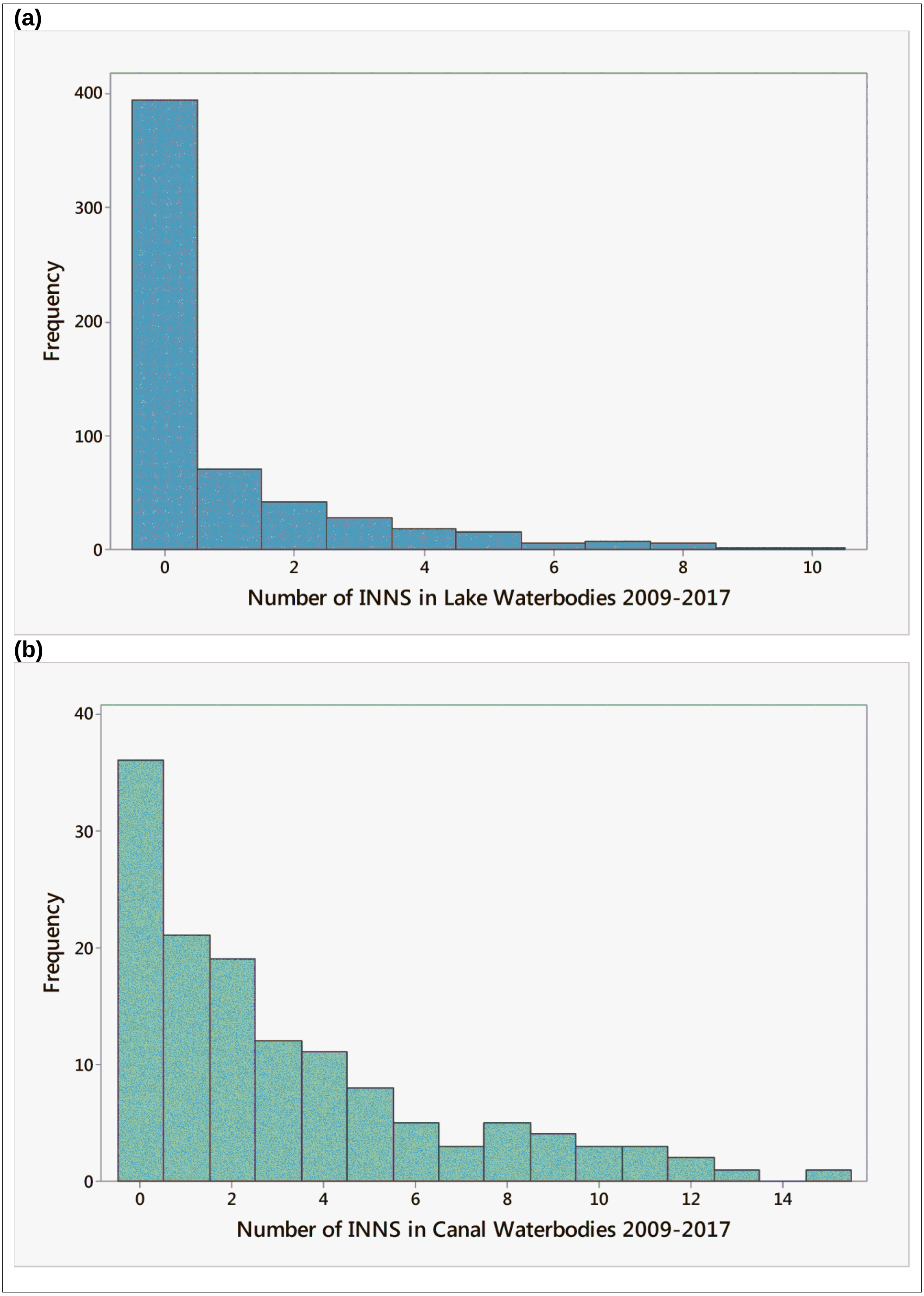


Figure 3.3 Frequency of INNS recorded 2009 to 2017 in (a) lake water bodies and (b) canal water bodies

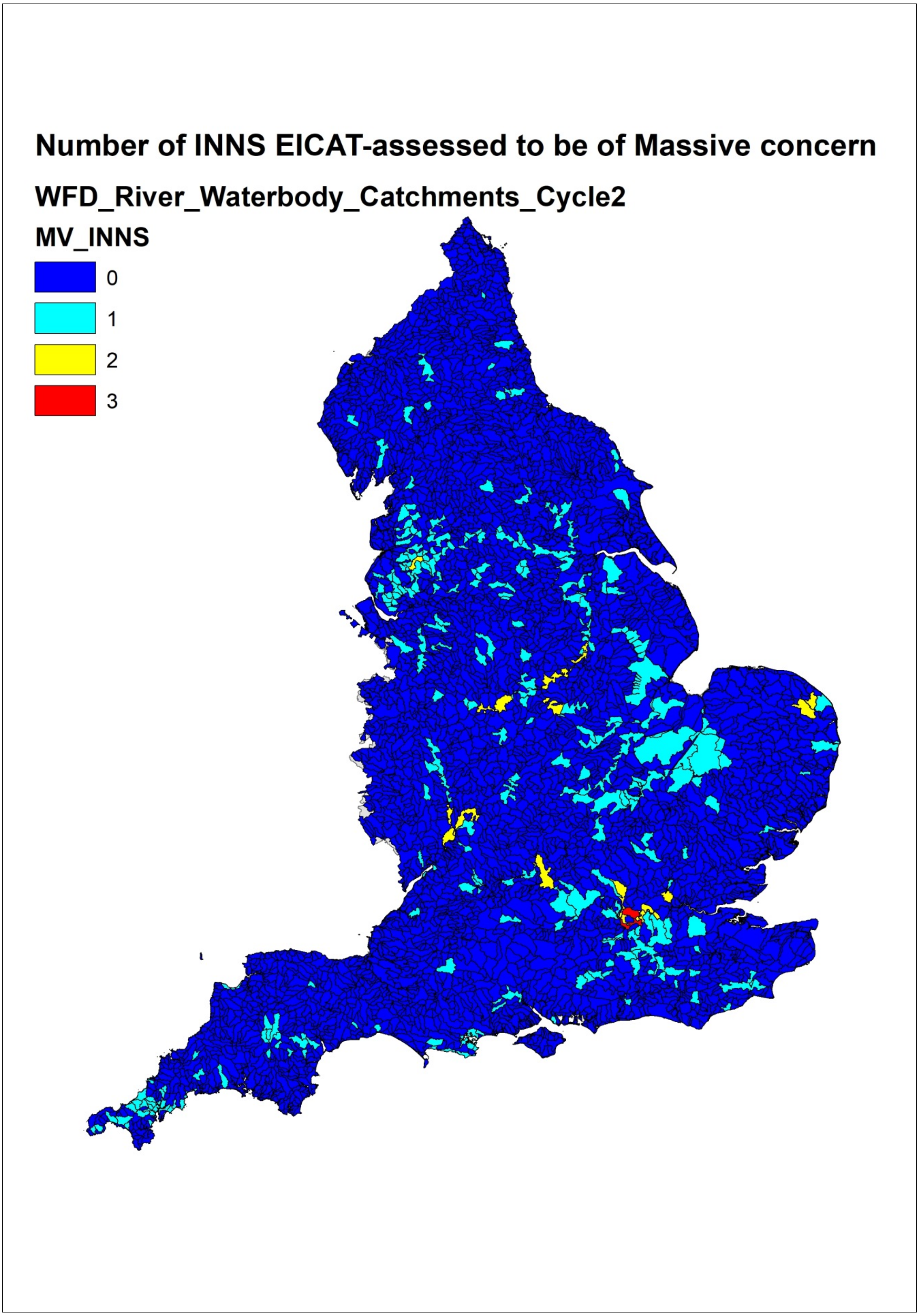


Figure 3.4 Number of INNS EICAT-graded as MV recorded 2009 to 2017 per river water body catchment (WFD cycle 2)

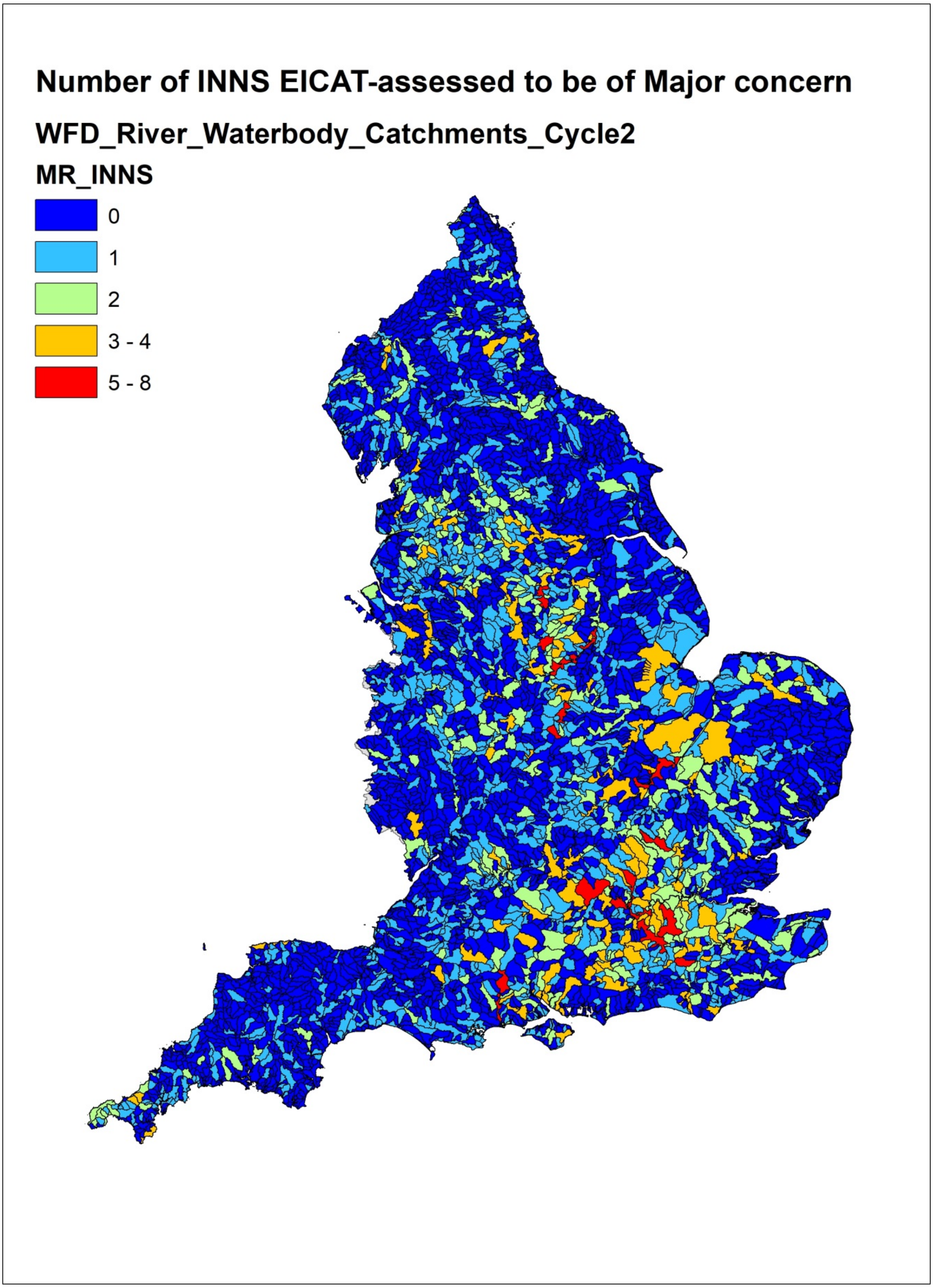


Figure 3.5 Number of INNS EICAT-graded as MR recorded 2009 to 2017 per river water body catchment (WFD cycle 2)

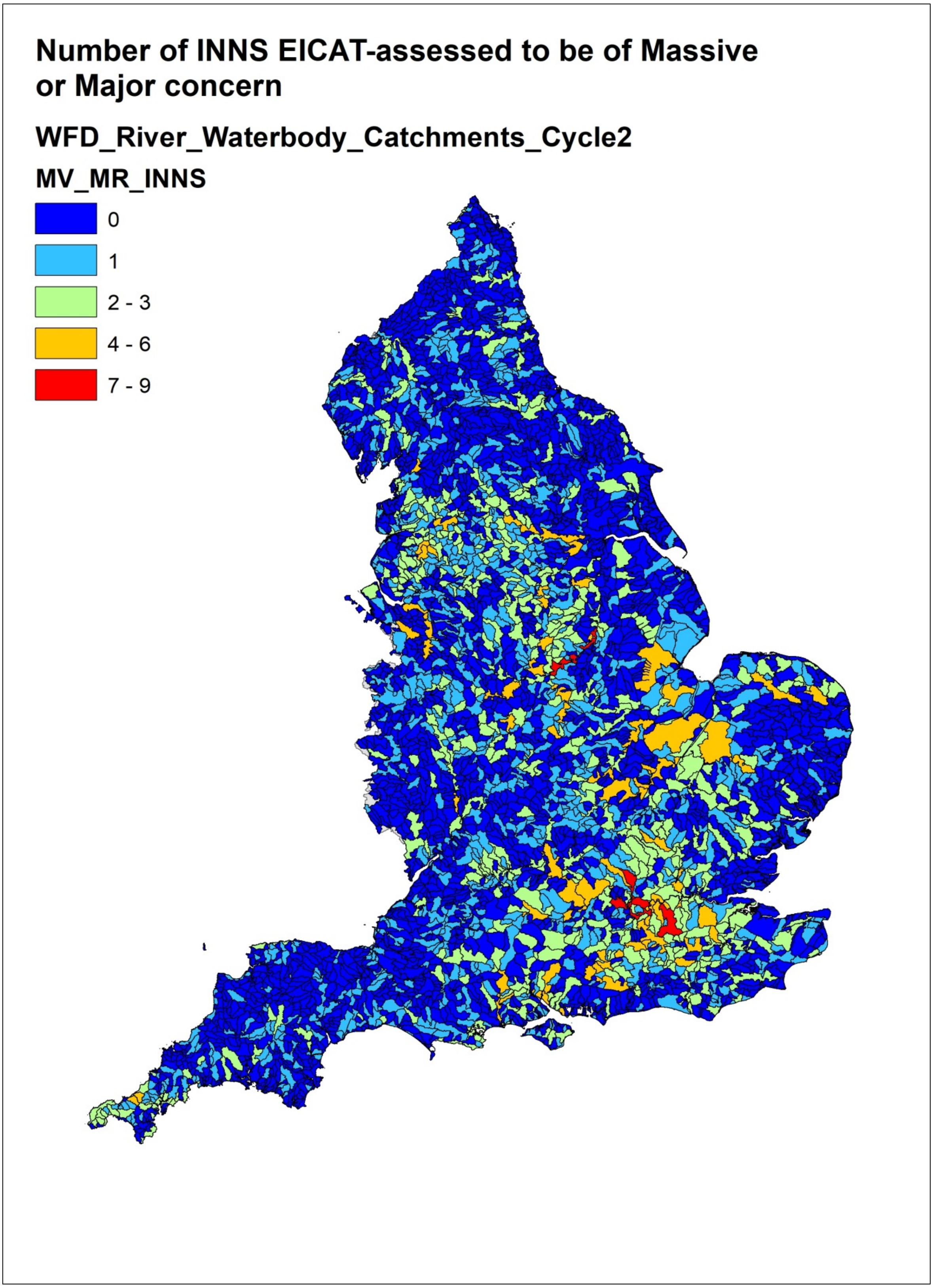


Figure 3.6 Number of INNS EICAT-graded as MV or MR (combined) recorded 2009 to 2017 per river water body catchment (WFD cycle 2)

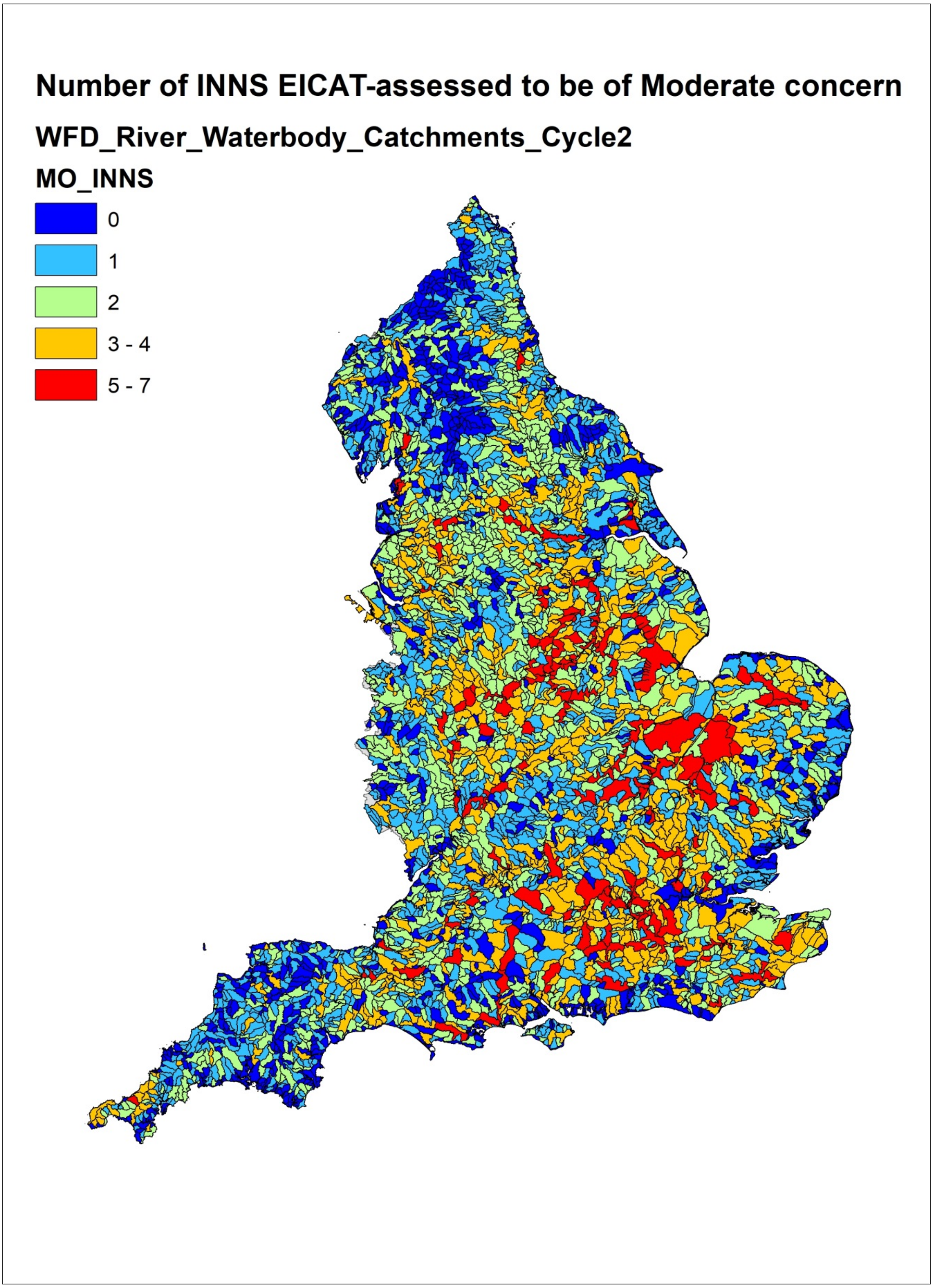


Figure 3.7 Number of INNS EICAT-graded as MO recorded 2009 to 2017 per river water body catchment (WFD cycle 2)

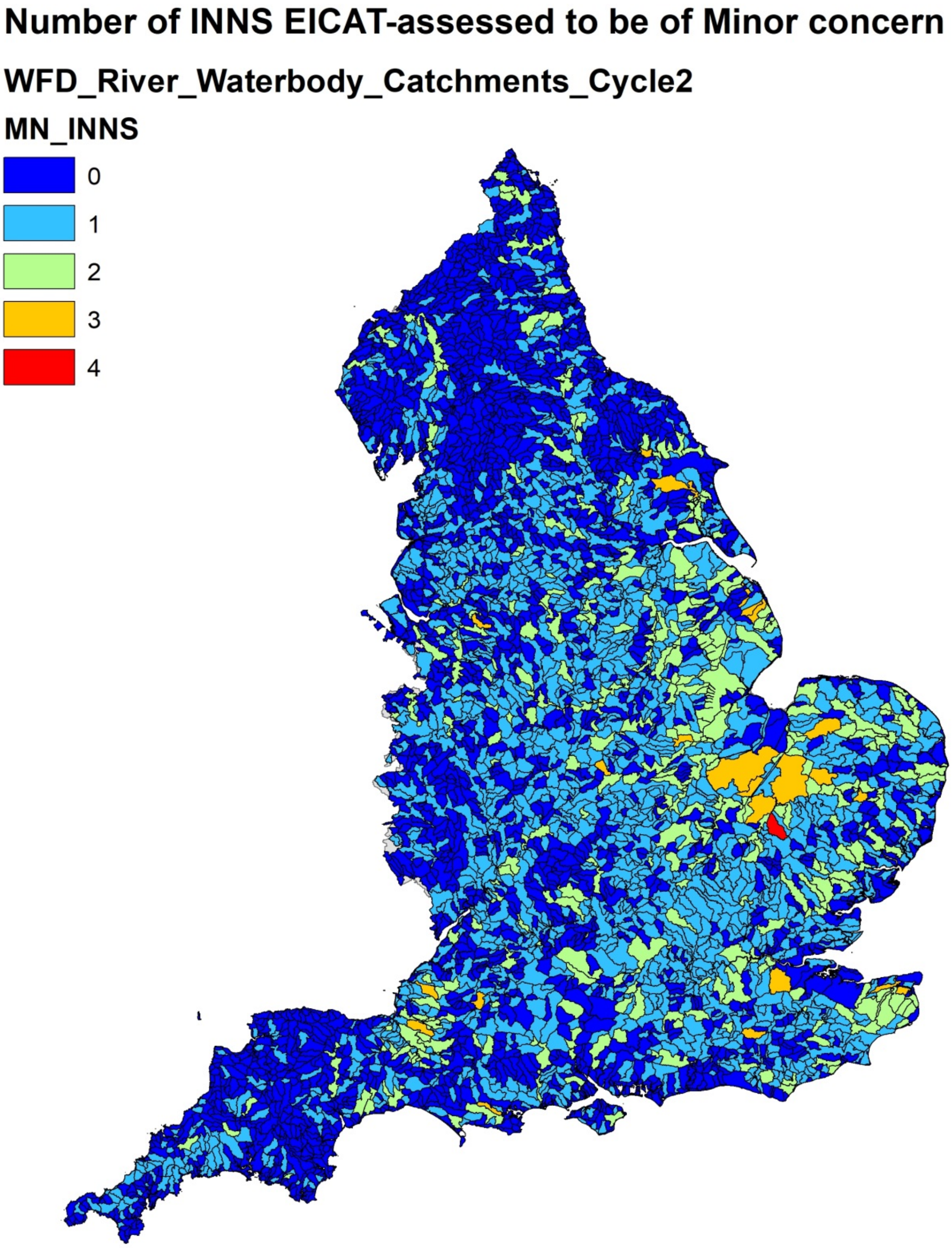


Figure 3.8 Number of INNS EICAT-graded as MN recorded 2009 to 2017 per river water body catchment (WFD cycle 2)

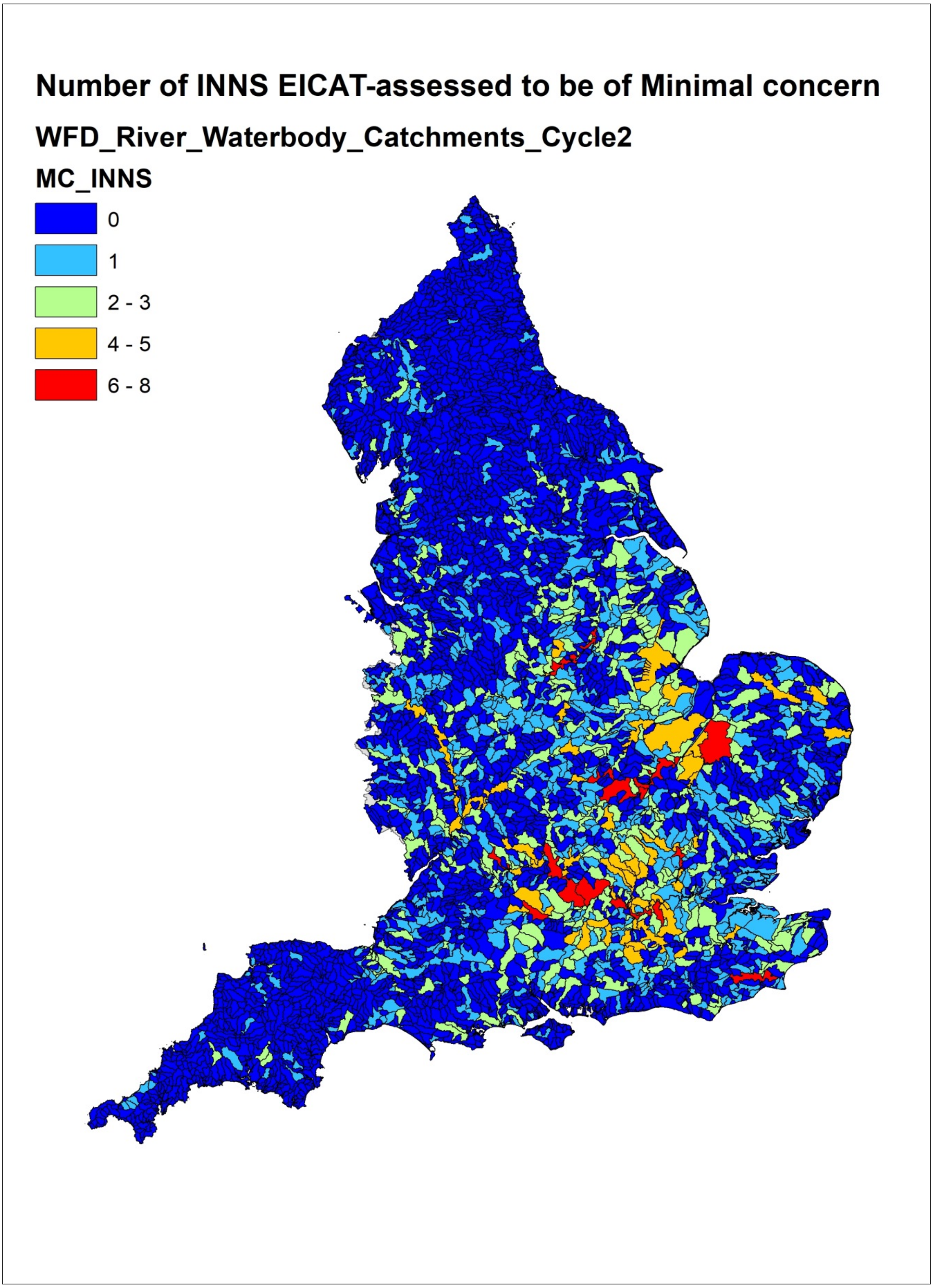


Figure 3.9 Number of INNS EICAT-graded as MC recorded 2009 to 2017 per river water body catchment (WFD cycle 2)

3.2 Estimating area of extent of individual species

To illustrate the current distribution of INNS relevant to the Environment Agency, data on species occurrences matched to water bodies were used to determine the area of extent for each species listed in Table 2.1.

Due to uncertainties in identification, all records of *Mimulus guttatus, Mimulus luteus* and *Mimulus guttatus x luteus* were treated as *Mimulus guttatus/luteus* group. Similarly, all records of *Physella acuta* and *Physella gyrina* were treated as *Physella.* Furthermore, there were no records of *Mimulus ringens* or *Myriophyllum heterophyllum* in the dataset. This meant that a total of 80 species (including 2 species groups) were considered.

The process used to calculate the area of extent was based on the approach used to develop the vascular plant Red List for England (Stroh et al. 2014). Three different methods were used to determine extent of occurrence:

- determining the number of distinct 10km squares in which the species had been recorded and then calculating the total area of these 10km squares
- minimum convex polygon (MCP)
- alpha hull

All 3 approaches were used as each has distinct benefits and drawbacks with respect to estimating the distribution of water bodies that are potentially at risk from each INNS.

The first of these methods involves determining, for each decade, the number of distinct OS 10km squares that contained occurrence data for each species and then the total area obtained by summing the area of these 10km squares. This method relies only on sites where records have been confirmed for the species, and does not include those areas where the species may be present and not recorded, and those suitable areas within its range where the species has yet to reach. Although this provides an estimate of the area of extent with the most confidence that the species was present throughout the area, it is the minimum potential area of extent.

The second method involved calculating the MCP for the occurrence data for each decade. The MCP is defined as the smallest polygon in which no internal angle exceeds 180° and which contains all the sites of occurrence. The MCP is often criticised due to a tendency for non-suitable areas to be included within the shape, especially when fitted to areas in which there are large areas of unsuitable habitat surrounded by suitable habitat. To reduce the extent to which this problem affected the areas estimated, a secondary polygon was created by intersecting the MCP polygon with the land mass of England (or a 50km wide coastal region in the case of marine species) and the area (in km^2 and also as a percentage of the total land/marine buffer area of England) of this polygon was then calculated.

The third method fitted an alpha hull to the occurrence data for each decade. Alpha hulls are a generalisation of the convex polygon and have been suggested to be more suitable to species distributions than the MCP, especially for irregularly shaped species ranges (Burgman and Fox 2003). Alpha hulls are created by a Delaunay triangulation of the data points (joining all points so that no lines intersect between points) and then selectively removing lines from this triangulation based on the value of a parameter α. The smaller the value of α, the finer the resolution of the hull produced. As α increases, the alpha hull will approach the MCP. There is no ideal value of α; instead the choice depends on the quality of the data and the aims of the study. For the indicator analyses, an α value of 80,000 was used. To minimise the inclusion of unsuitable habitat (at the scale of resolution used), the alpha hull was also intersected with the land mass of England to produce a new hull for which the area (in km^2 and as a percentage of the total land area of England) was calculated.

The areas of extent calculated by all 3 methods for all records and each decade are provided in Appendix A for each INNS considered.

A series of maps and area estimates was produced for each INNS from these analyses based on all records. Figure 3.10 shows an example of the maps produced, in this case for *Pacifastacus leniusculus* (signal crayfish). All the other maps are provided in Appendix B.

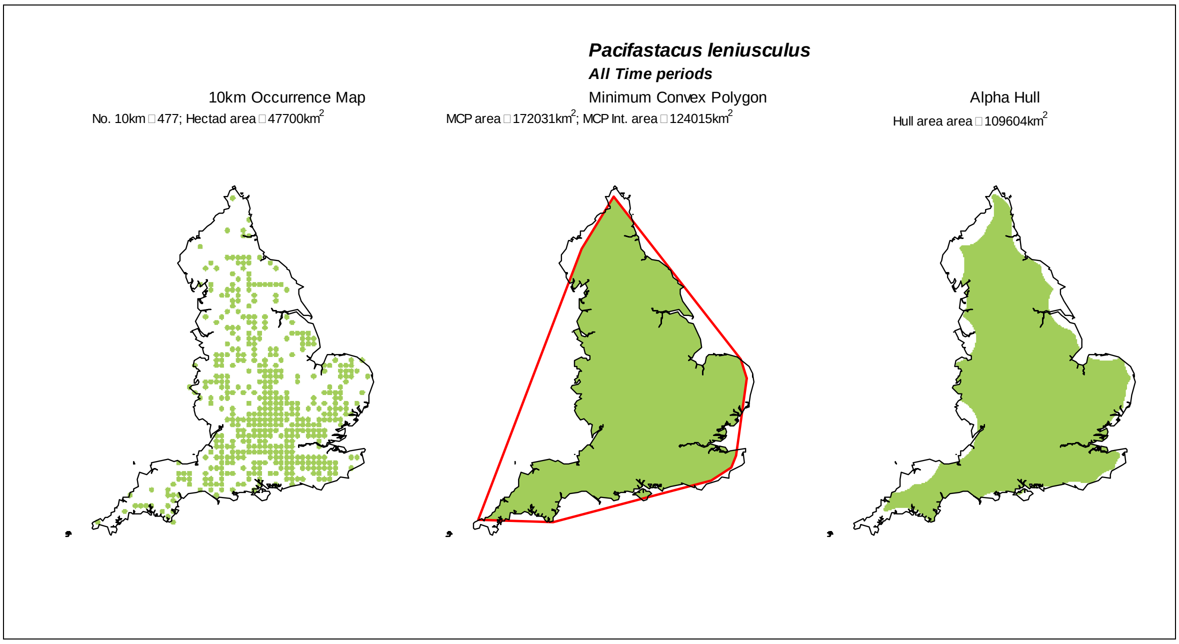
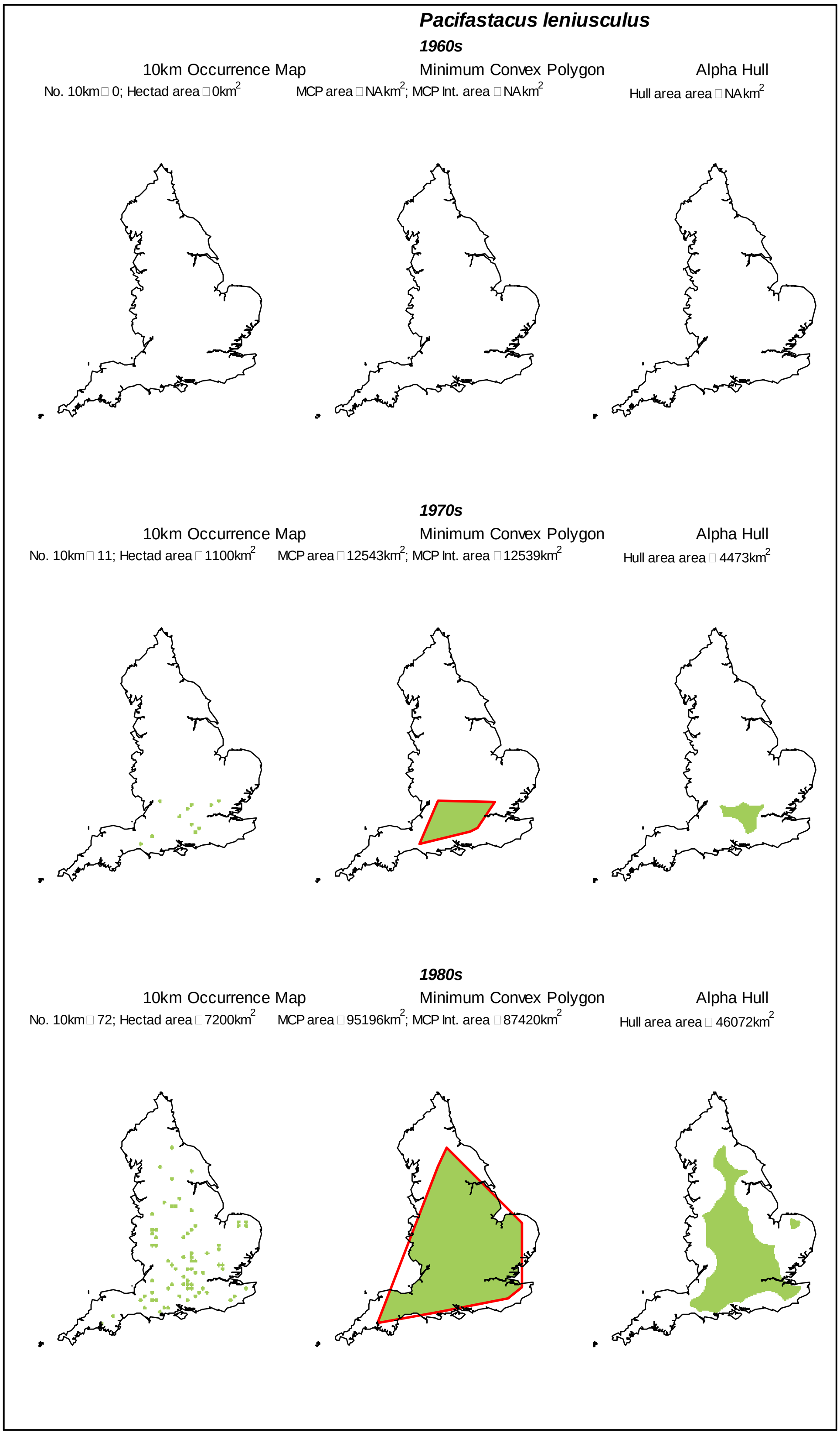


Figure 3.10 Example area of extent maps for *Pacifasticus leniusculus* (signal crayfish) using all records.

Notes: The first map shows the 10km occurrence data, the second map shows the MCP (outlined by a red line) and its intersection with the land (green filled region), and the third map shows the alpha hull and its intersection with the land (green filled region).
The labels above each map give the total area of distinct 10km squares, the area of the MCP/England land intersection and the area of the alpha hull/England land intersection respectively.

To visualise the spread of individual INNS, the data were divided into decades according to the date that each record of the species was made, and maps and area estimates were produced using these data. Figure 3.11 shows an example of the maps produced – again for *Pacifastacus leniusculus* (signal crayfish), illustrating the spread from early introduction. All other maps are provided in Appendix C.

(a)


Pacifastacus leniusculus
1960s
10km Occurrence Map
No. 10km□ 0; Hectad area □0km^2
Minimum Convex Polygon
MCP area □NA km^2; MCP Int. area □NA km^2
Alpha Hull
Hull area area □NA km^2
1970s
10km Occurrence Map
No. 10km□ 11; Hectad area □1100km^2
Minimum Convex Polygon
MCP area □12543km^2; MCP Int. area □12539km^2
Alpha Hull
Hull area area □ 4473km^2
1980s
10km Occurrence Map
No. 10km□ 72; Hectad area □7200km^2
Minimum Convex Polygon
MCP area □95196km^2; MCP Int. area □87420km^2
Alpha Hull
Hull area area □ 46072km^2

(b)

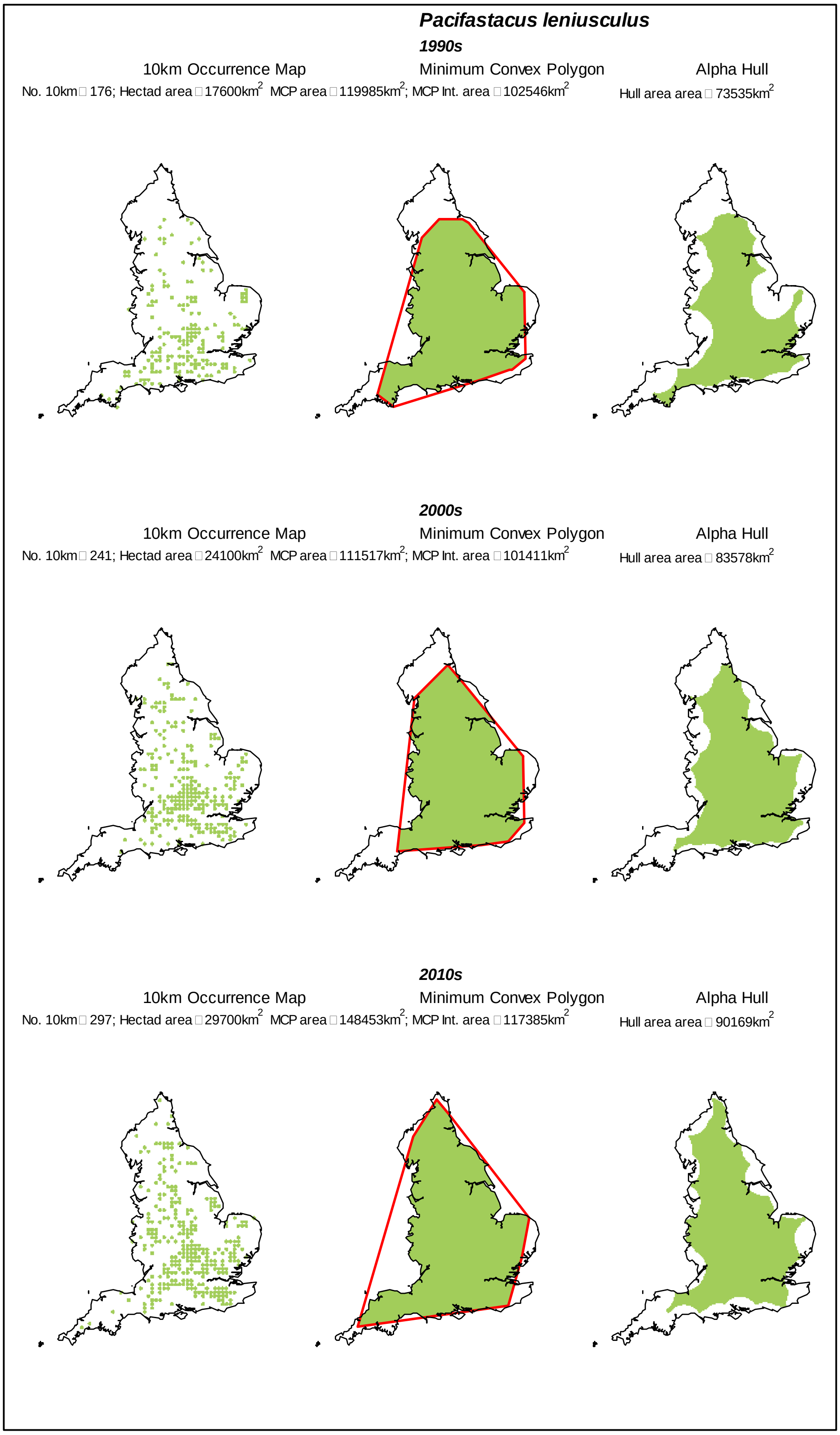


Figure 3.11 Change in area of extent for *Pacifasticus leniusculus* (signal crayfish) by decade (1960s to 2010s)

Notes: For each decade, the first map shows the 10km occurrence data, the second map shows the MCP (outlined by a red line) and its intersection with the land (green filled region), and the third map shows the alpha hull and its intersection with the land (green filled region).

4 INNS and measures of ecological quality

4.1 Introduction

The project's first objective was to determine if the presence of INNS had a significant influence on ecological quality as measured by the WFD tools. These tools return an EQR based on the measure of the biological quality element (BQE) (macroinvertebrates, macrophytes, fish) observed at the test site compared with the BQE measure expected if the site was under reference conditions.

EQR is the measure of ecological quality used to classify the ecological status of sites according to the WFD. Any significant impact of INNS on EQR will influence the ecological status of the water body and have an impact on WFD objectives. Statistical analyses therefore focused on those samples for which EQR data were available with the aim of detecting any significant impact of INNS on EQR.

The impacts of INNS are not restricted to those on functionally equivalent native species (that is, impacts within the same BQE as the INNS). They have the potential to cause changes that propagate through ecosystems, affecting multiple components of the community (that is, impacts on BQEs other than that of the INNS). It was therefore important to ensure that data for potential effects on all BQEs were analysed where possible.

Statistical tests were conducted to detect any difference in EQR between sites that could be attributed to the presence of INNS (see Section 4.3).

4.2 Sampling units

As the water bodies contained multiple sampling sites and had been sampled for different reasons at different times, the likelihood of individual sites where INNS occurred being sampled repeatedly for different BQEs over time was low. This had implications for:

- the type of statistical analyses possible
- the level of confidence that the INNS was present or absent at the site when the sample used to derive the EQR was collected
- the inferences that could be drawn from any statistically significant result

The data that could be used to relate the presence of INNS to EQR as measured by the WFD tools were therefore extracted from the Microsoft SQL database at 2 scales:

- reach year (see Section 4.2.1)
- water body reporting period (see Section 4.2.2)

Due to the important differences between the 2 datasets, the statistical approaches used to analyse them, and the inferences drawn from the results, also differed.

4.2.1 Reach scale over individual years

Here the river or canal reach was regarded as the sampling unit. Samples collected from the same reach (as matched by GIS; see Section 2.3) within the same year as that where the INNS was recorded as present (or absent for the control group) were used. Reaches where the INNS was recorded at any time were excluded from the control group.

These data provided a high confidence that the INNS was present/absent at the site at the time of sampling. However, these data were less likely to include information on multiple BQEs – as other reaches within the water body may have been sampled for other BQEs or the reach sampled in different years. Furthermore, at this scale there was a lower probability of individual reaches being repeatedly sampled over time.

Statistical tests using these data were able to identify differences in EQR among reach years that were associated with the presence of the INNS, but cannot categorically attribute any causality to the INNS.

4.2.2 Water body scale over WFD reporting periods

Here the WFD water body was regarded as the sampling unit. Samples collected from the same water body within the same 3-year WFD reporting period as that where the INNS was recorded as present (or absent for the control group) were used. Water bodies where the INNS was absent were excluded from the control group if the INNS had been recorded as present in the water body previously.

As it was possible that an INNS may be in a river/canal/lake water body yet not be recorded at the location where the sample used to derive EQR was collected, the confidence that the INNS was present/absent at the site at the time of sampling was lower (that is, a higher probability of a false positive or false negative). However, these data were more likely to include information on multiple BQEs and provided more potential for detecting wider ecosystem impacts of INNS. Furthermore, at this scale there was a higher probability of individual water bodies containing INNS being repeatedly sampled over time, providing greater confidence when attributing causality to any observed difference in EQR associated with the presence of INNS.

4.3 Data analysis

4.3.1 Reach year scale over individual years

Data extracted at the reach year scale were classified into 2 groups:

- those reach years where the INNS had been recorded
- a control group of reach years covering the same time periods from similar sites (as determined by the WFD System A typology) where the INNS was recorded as being absent (that is, a sample of the BQE containing the INNS was collected but the INNS was not recorded)

These data were analysed using 3 statistical tests (paired t-test, logistic regression and quantile regression) to investigate different aspects of any difference detected between reach years where the INNS was present and those where it was absent. See below for details of the 3 tests.

These tests were carried out to determine if the presence of an INNS was associated with a significant difference in EQR compared with the control group (that is, reach years from similar water bodies). Logistic regression and quantile regression are less vulnerable to the influence of other factors on EQR as they concentrate on the upper limit and address the question: ‘Is EQR constrained to lower values where INNS are present?’. However, the strongest evidence is provided when all 3 tests coincide.

Due to the nature of the data used in these tests, it was not possible to confidently attribute any difference detected to the presence of the INNS. At this scale, the results indicate an association between the presence of the INNS and the returned EQR (that is, the INNS may be present because the ecological quality of the site differed from the control group rather than causing it to differ).

Paired t-test

This test was used to determine if the EQR from reach years that contained the INNS was significantly different to the population of EQR values returned for the control group reach years where the INNS was absent. This test considered the mean and the whole distribution of EQR values from the 2 groups.

H_1 = There is a significant difference between the mean EQR of reach years where the INNS was present and those where it is absent.

Logistic regression

This test was used to determine the influence of the presence of the INNS on the upper limit of EQR. Here it was assumed that other factors may influence the EQR from reach years where the INNS was present such that they may return a low EQR, but that the INNS, if having a significant influence, would inhibit the reach year from achieving a high EQR. This meant that reach years where the INNS was present would be associated with a lower EQR.

H_1 = There is a higher probability of the INNS being absent from reach years with a high EQR.

Quantile regression

This test also compared the upper limit and was used to determine if the highest 10% (90th quantile: Q_{90}) of EQR values from reach years where the INNS was present was lower than that from reach years where it was absent.

H_1 = There is a significant difference between the highest 10% of EQR from reach years where the INNS is present and those where it is absent.

4.3.2 Water body scale over WFD reporting periods

The data extracted from the SQL server database consisted of:

- those water bodies where the INNS had been recorded during that WFD reporting period
- those water bodies where the INNS was not recorded as being present throughout that reporting period

Care was taken to ensure consistency in water body identities over WFD reporting cycles where changes had occurred.

These data were screened further to identify those water bodies where:

- the INNS was recorded as being absent early in the time series (including before implementation of the WFD) but present during later reporting periods
- there were measures of EQR during both the period when the INNS was absent and when it was present

In all cases, the first WFD reporting period with an EQR had to be associated with the absence of the INNS and all samples from that water body prior to the first EQR indicated the absence of the INNS.

Data were discounted where the water body periods did not follow the logical sequence of INNS absent followed by INNS present. Thus, a set of ‘impacted’ WFD water bodies was defined for each INNS where the EQR followed a time series of before and after the occurrence of the INNS.

For each INNS, a ‘control’ group of WFD water bodies was drawn from physicochemically similar water bodies (as determined by the WFD System A typology) with an EQR from the same region of the country and covering the same time periods as those of the ‘impacted’ group, but where the INNS was not detected within the water body throughout the time series. The use of an adequate ‘control’ group is fundamental for attribution of any causal effect to the ‘impact’ (invasion by the INNS) rather than a temporal trend (Underwood 1994).

Thus, the datasets for analysis comprised ‘before’ and ‘after’ time periods for both the impacted and control group of WFD water bodies. Data were further divided into those WFD water bodies where the INNS first occurred during the following WFD reporting time periods:

- early (2007 to 2009)
- middle (2010 to 2012)
- late (2013 to 2015)

Note that only data from 2013 and 2014 were available in the late WFD reporting period.

Data were analysed using asymmetric analysis of variance following a before–after–control–impact (BACI) design with multiple water bodies within each of the impacted and control groups, multiple sites nested within water bodies and sites/water bodies sampled over multiple years, nested within both the before and after time periods (where possible).

By including replicate water bodies within the impacted group, the design of this test substantially reduced the probability of committing a Type II error common in BACI designs (Underwood 1994). Here, a significant interaction between time (before/after) and the presence of the INNS (control/impact) indicated an impact of the INNS on EQR, although the probability of detecting an impact is dependent on the within-subject replication and the effect size.

As detecting press impacts (such as the invasion of a site by an INNS) requires maximal numbers of control locations (Underwood and Chapman 2003), care was taken to ensure that an adequate number of control water bodies were included in each test. Where multiple tests were undertaken for each measure of EQR (as a consequence of dividing the data into those WFD water bodies where the INNS first occurred during early, middle or late WFD reporting time periods), a Bonferroni correction was applied to the *p* level accepted as significant.

These tests were made to determine if the occurrence of an INNS was associated with a significant change in EQR when compared with the control group (that is, the response over time for water body periods from similar water bodies). Due to the requirements of the data for these tests, those INNS that have expanded their range substantially during the time period for which EQR data were available provided the largest datasets, and thus the highest likelihood of detecting an impact.

There are some constraints on the data used for these tests due to the assumptions made about:

- the presence or absence of the INNS
- the differences between the control and impacted group of WFD water bodies

However, the BACI design provides strong evidence that the INNS is responsible for any change in EQR over time.

As the water bodies were allocated to treatments, there are also assumptions made about how representative individual impacted (and control) sites are. This was particularly with regard to the effect of the INNS relative to other factors influencing the variation in EQR over time. With higher replication, the influence of these assumptions becomes less apparent in both the impacted and control groups of water bodies.

The strongest evidence for an impact of the INNS on the measures of ecological quality is obtained when the results of analysis made at reach year and water body period scales coincide.

4.4 Results

4.4.1 Reach scale over individual years

Data were extracted at the reach year scale and statistical tests of their impact on EQR were made for the following INNS:

- zebra mussel (*Dreissena polymorpha*), MV
- signal crayfish (*Pacifastacus leniusculus*), MR
- floating pennywort (*Hydrocotyle ranunculoides*), MR
- common carp (*Cyprinus carpio*), MR
- demon shrimp (*Dikerogammarus haemobaphes*), MO
- Nuttall's pondweed (*Elodea nuttallii*), MO
- least duckweed (*Lemna minuta*), MO
- giant hogweed (*Heracleum mantegazzianum*), MO
- Himalayan balsam (*Impatiens glandulifera*), MO
- zander (*Sander lucioperca*), MO
- sunbleak (*Leucaspius delineatus*), MN

These species were selected during the data screening exercise according to the likely availability of data and their perceived threat (including grades of concern for species in Great Britain allocated to them by the GBNNSS).

For each INNS where sufficient data were available, the impact on 4 measures of EQR derived from 3 BQEs was tested. These were:

- Macrophyte EQR – as derived by the LEAFPACS tool (WFD UKTAG 2014b)
- Fish EQR – as derived by the FCS2 tool (WFD UKTAG 2008)
- NTAXA EQR – as derived by the RICT tool (WFD UKTAG 2014a)
- ASPT EQR

Differences in the values of the EQR used to define the classification boundaries mean that these 4 measures of EQR need to be treated separately. In each case, however, a lower EQR corresponds with lower ecological quality.

Any change in measured ecological quality due to the impacts of INNS will be reflected in the EQR returned by the WFD tools.

Sufficient data to undertake robust analyses were not available for all measures of EQR for some species.

Zebra mussel (Dreissena polymorpha)

There were insufficient data to test the effects on EQR of macrophytes; the effects on the other 3 measures of EQR are shown in Figure 4.1. For the other measures of EQR, the only significant result was a lower 90th percentile of EQR of ASPT in reach years where zebra mussel was present (Table 4.1).

Given that zebra mussels have been given an EICAT classification of massive concern, it is surprising that a more substantial result was not found. However, a limited amount of data was available from lakes, where zebra mussels have been reported as having profound impacts. Furthermore, the density of zebra mussels at invaded sites was not included in the analysis and impacts are likely to be related to density.

Reach year scale analysis: zebra mussel (*Dreissena polymorpha*)

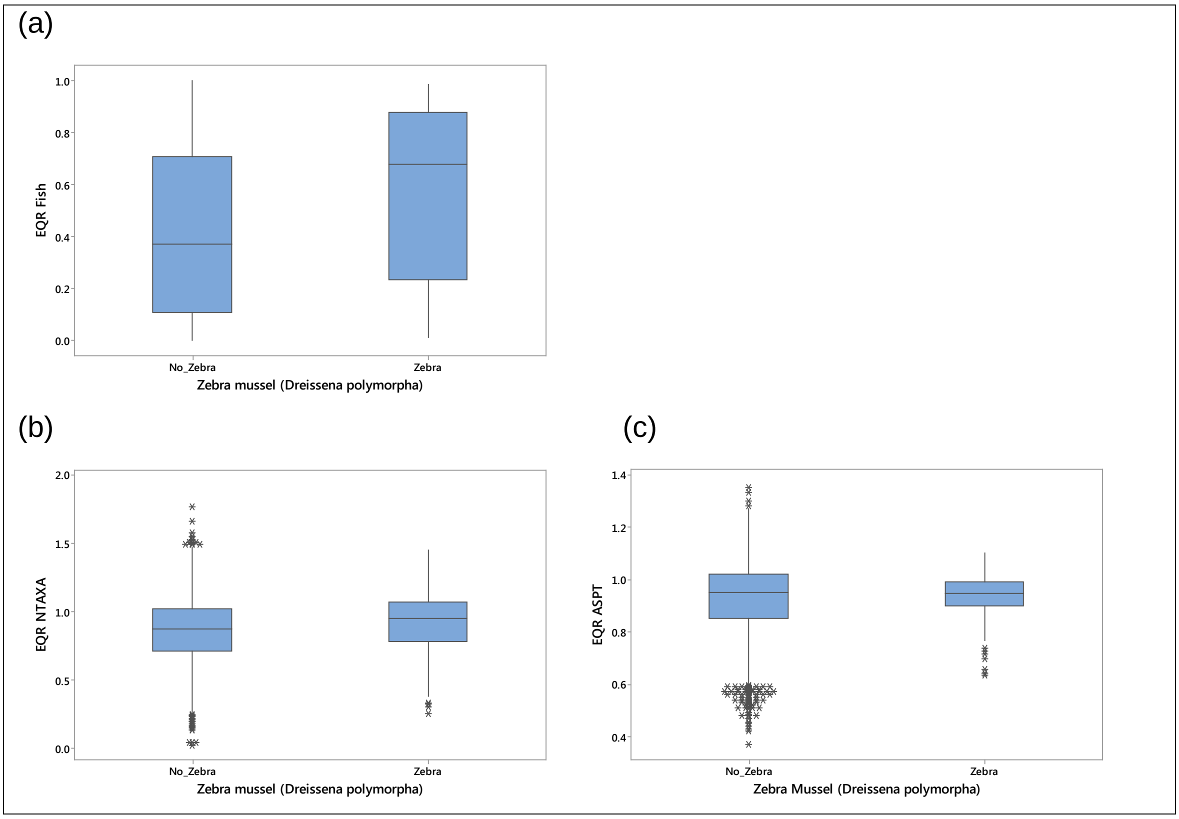


Figure 4.1 Box plots of EQR of (a) fish, and invertebrate (b) NTAXA and (c) ASPT from reach years with and without zebra mussel

Table 4.1 Results of statistical tests of the association between zebra mussel and EQR at the reach year scale

	N_1	N_0	Difference between means	*p*		
				t-test	Logistic	Q_{90}
EQR Macrophyte	1					
EQR Fish	10	424	+0.155	0.189	0.1446	0.6510
EQR NTAXA	119	7,214	+0.0367	0.096	0.0769	0.7373
EQR ASPT	119	7,214	–0.00054	0.949	0.9641	**<0.0001**

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Signal crayfish (Pacifastacus leniusculus)

Sufficient data were available to test all 4 measures of EQR (Figure 4.2).

All 3 statistical tests detected significant differences in the EQR of macrophytes, NTAXA and ASPT in reach years where signal crayfish were present (Table 4.2). The EQR of macrophytes and NTAXA were substantially lower (mean difference of –0.103 and –0.101 respectively) in the reach years with signal crayfish.

These differences are substantial relative to the width of the WFD classes for these measures of ecological quality (macrophytes class width = 0.2; NTAXA class width = 0.2–0.14). As signal crayfish are omnivorous, it is plausible that they may consume other invertebrates and plants leading to a decline in numbers of species of these 2 BQEs. Predation on the eggs and juveniles of fish by signal crayfish has also been reported, but no difference in EQR of fish was detected.

The EQR of ASPT was significantly higher (mean difference of +0.049 compared with a class width = 0.12–0.11) in reach years where signal crayfish were present. The Whalley, Hawkes, Paisley and Reigg (WHPT) scoring system for deriving ASPT explicitly includes signal crayfish, giving Astacidae (including non-native species) a relatively high score of 7.9 (WFD UKTAG 2014a). The Biological Monitoring Working Party (BMWP) system also included signal crayfish, giving them a score of 8.

Although it is plausible that signal crayfish target low scoring prey, resulting in their loss from invaded sites (Crawford et al. 2006, Mathers 2016a, Mathers 2017, Turley et al. 2017,), if signal crayfish cause other species to be extirpated from sites where they are present, due to their high score it is likely the average score (ASPT) would increase. Although non-native Astacidae are explicitly included in the scoring system for the WFD invertebrate tool, they are not alone. Many INNS are included when deriving EQR.

Reach year scale analysis: signal crayfish (*Pacifastacus leniusculus*)

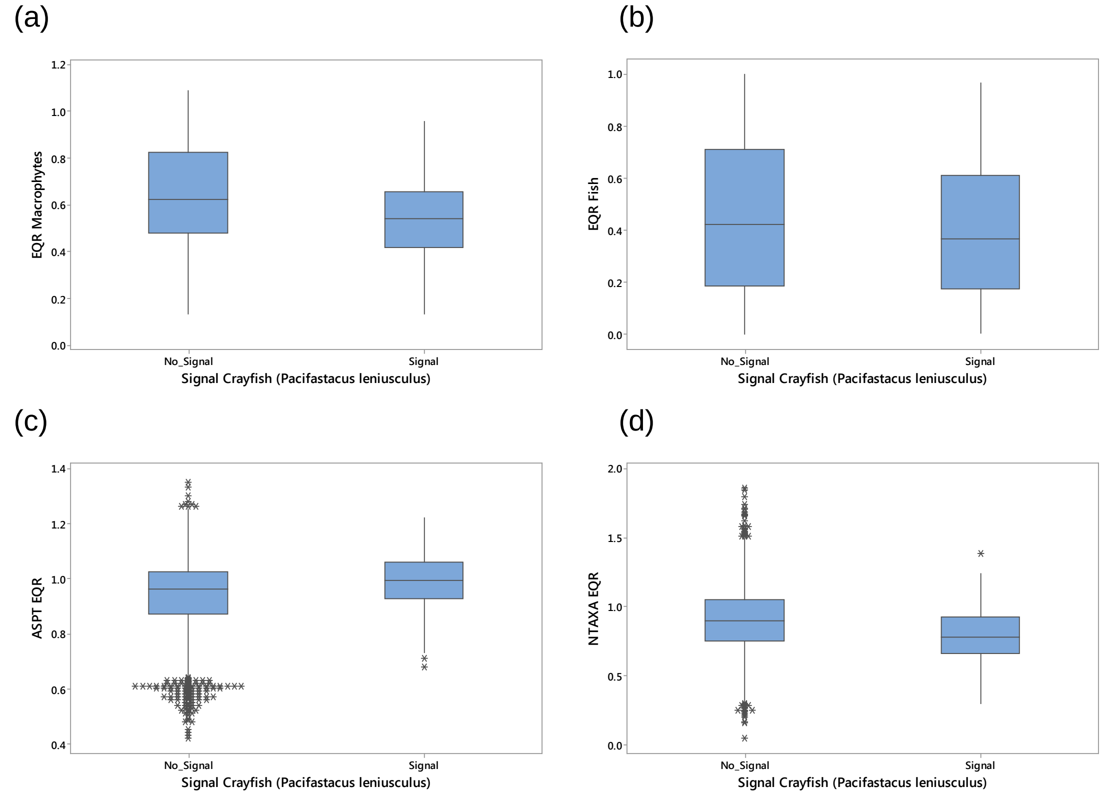


Figure 4.2 Box plots of EQR of (a) macrophytes, (b) fish, and invertebrate (c) NTAXA and (d) ASPT from reach years with and without signal crayfish

Table 4.2 Results of statistical tests of the association between signal crayfish and EQR at the reach year scale

	N_1	N_0	Difference between means	*p*		
				t-test	**Logistic**	**Q_{90}**
EQR Macrophyte	60	981	–0.103	**<0.001**	**<0.0001**	**0.0401**

	N_1	N_0	Difference between means	*p*		
				t-test	Logistic	Q_{90}
EQR Fish	48	155	–0.056	0.240	0.257	0.5281
EQR NTAXA	434	7,277	–0.101	**<0.001**	**<0.0001**	**<0.0001**
EQR ASPT	434	7,277	+0.049	**<0.001**	**<0.0001**	**<0.0001**

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Floating pennywort (Hydrocotyle ranunculoides)

Sufficient data were available to test all 4 measures of EQR (Figure 4.3). All 3 statistical tests detected significant differences in EQR of ASPT in reach years where floating pennywort was present and 2 of the 3 tests detected a difference in EQR of macrophytes, where quantile regression was marginally not significant (Table 4.3). In both cases, EQR was lower in reach years where floating pennywort was present.

As floating pennywort grows over the surface of water bodies, it could reduce light and oxygen in the water below, where it forms a thick carpet.

Reach year scale analysis: floating pennywort (*Hydrocotyle ranunculoides*)

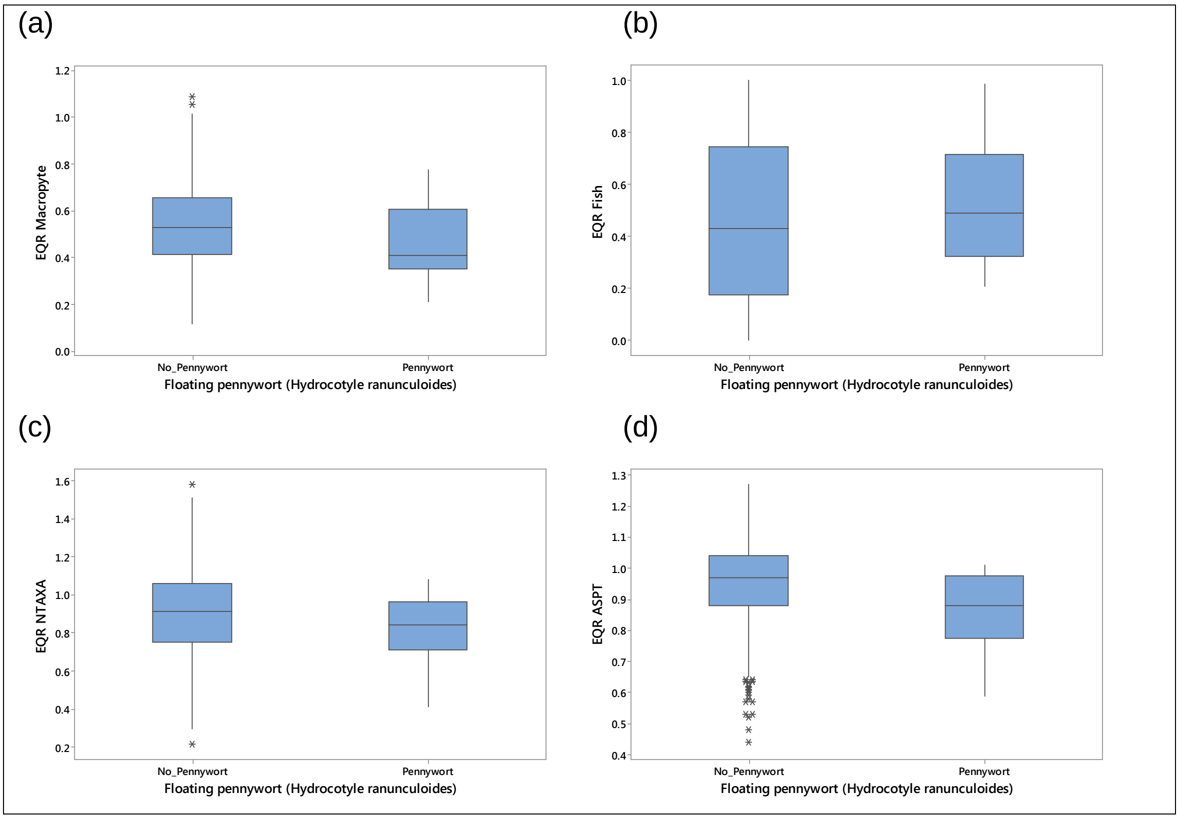


Figure 4.3 Box plots of EQR of (a) macrophytes, (b) fish, and invertebrate (c) NTAXA and (d) ASPT from reach years with and without floating pennywort

Table 4.3 Results of statistical tests of the association between floating pennywort and EQR at the reach year scale

	N_1	N_0	Difference between means	*p*		
				t-test	**Logistic**	**Q_{90}**
EQR Macrophyte	24	3,232	–0.0893	**0.007**	**0.0170**	0.0600
EQR Fish	9	200	+0.0583	0.524	0.5792	0.7884
EQR NTAXA	17	926	–0.0841	0.078	0.1319	0.1620
EQR ASPT	17	926	–0.0945	**0.008**	**0.0027**	**0.0002**

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Common carp (Cyprinus carpio)

Sufficient data were only available to test the influence of common carp on EQR of fish (Figure 4.4). All 3 statistical tests were not significant, although the t-test and logistic regression were close to significance.

Common carp is one of the 23 species of fish included in the Fish Classification Tool (WFD UKTAG 2008). If common carp are found at a site where they are expected, a high EQR is returned, whereas if they are found where they are not expected (and vice versa), a low EQR is returned. Hence it is difficult to predict how the EQR of fish would respond to the presence of common carp. Unfortunately, there were insufficient EQR data on other BQEs to determine any effects of common carp as the types of sites where common carp were typically found were not often sampled for other BQEs. A limited amount of data was available from lakes, where common carp have been reported as having profound impacts.

Reach year scale analysis: common carp (*Cyprinus carpio*)

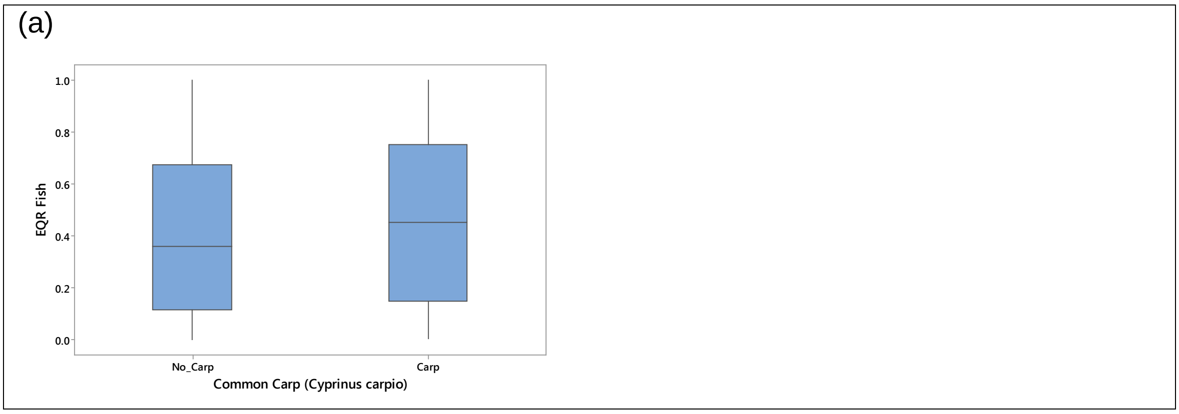


Figure 4.4 Box plots of EQR of (a) fish from reach years with and without common carp

Table 4.4 Results of statistical tests of the association between common carp and EQR at the reach year scale

	N_1	N_0	Difference between means	p		
				t-test	Logistic	Q_{90}
EQR Macrophyte	1					
EQR Fish	153	4,939	+0.0489	0.071	0.0601	0.2839
EQR NTAXA	6					
EQR ASPT	6					

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Demon shrimp (Dikerogammarus haemobaphes)

Sufficient data were available to test any association between demon shrimp and the EQR of macrophytes, NTAXA and ASPT (Figure 4.5). The t-test and logistic regression statistical tests returned a significant difference in EQR of ASPT, whereas quantile regression was not significant (Table 4.5), reflecting the fact that reach years where demon shrimp were found did not have a lower EQR for ASPT.

Reach year scale analysis: demon shrimp (*Dikerogammarus haemobaphes*)

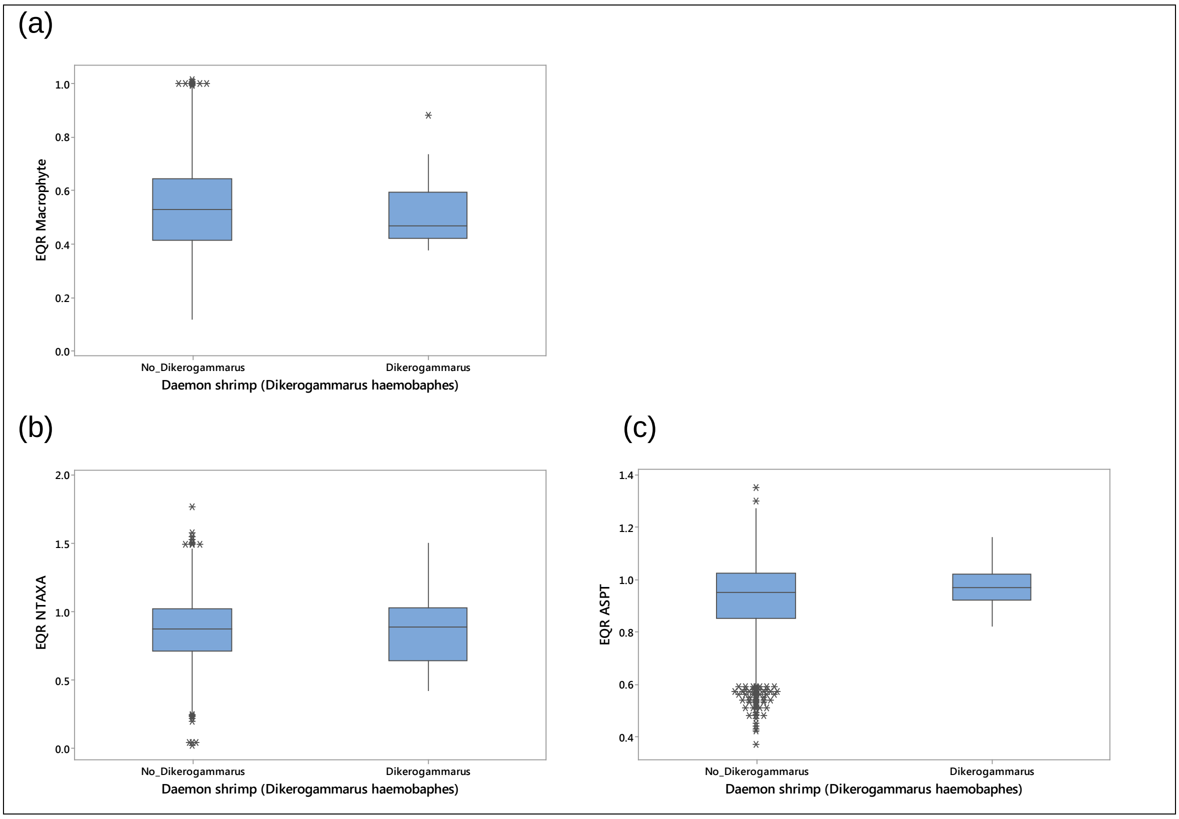


Figure 4.5 Box plots of EQR of (a) macrophytes, and invertebrate (b) NTAXA and (c) ASPT from reach years with and without demon shrimp

Table 4.5 Results of statistical tests of the association between demon shrimp and EQR at the reach year scale

	N_1	N_0	Difference between means	*p*		
				t-test	Logistic	Q_{90}
EQR Macrophyte	10	708	–0.0168	0.749	0.7658	0.7614
EQR Fish	0					
EQR NTAXA	48	5,922	–0.0159	0.641	0.6171	0.6100
EQR ASPT	48	5,922	+0.0433	**<0.001**	**0.0253**	1.0000

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Nuttall's pondweed (Elodea nuttallii)

Sufficient data were available to test all 4 measures of EQR (Figure 4.6). A significant difference in the EQR of macrophytes and ASPT was found for reach years where Nuttall's pondweed was present (Table 4.6), with all 3 statistical tests indicating a significant difference.

The mean difference in EQR of macrophytes was reasonably large (–0.0833), whereas the difference for ASPT was small (–0.02864) although significant.

As Nuttall's pondweed is included in the taxa considered by the macrophyte tool (River LEAFPACS2) with a River Macrophyte Nutrient Index (RMNI) score of 9.44 indicative of high nutrient conditions (WFD UKTAG 2014b), it is not clear if any influence on EQR of macrophytes was due to a real biological interaction.

Reach year scale analysis: Nuttall's pondweed (*Elodea nuttallii*)

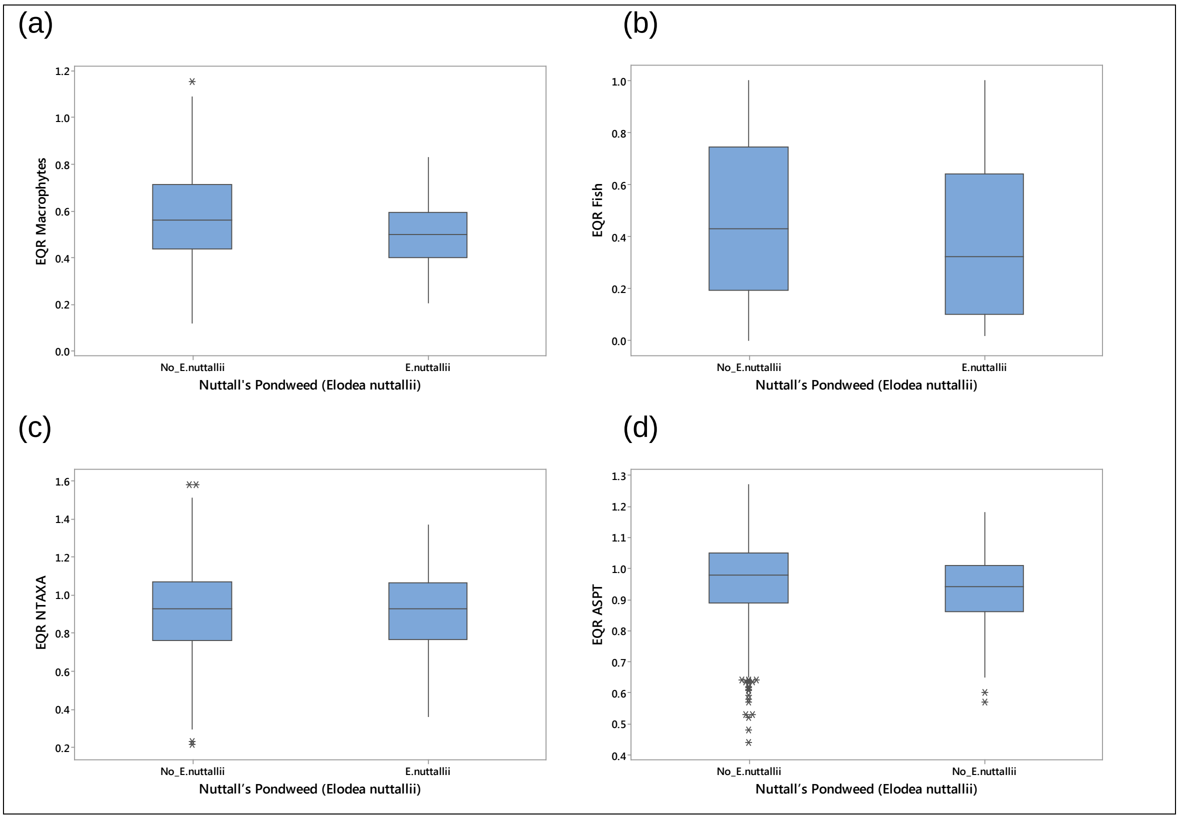


Figure 4.6 Box plots of EQR of (a) macrophytes, (b) fish, and invertebrate (c) NTAXA and (d) ASPT from reach years with and without Nuttall's pondweed

Table 4.6 Results of statistical tests of the association between Nuttall's pondweed and EQR at the reach year scale

	N_1	N_0	Difference between means	*p*		
				t-test	Logistic	Q_{90}
EQR Macrophyte	147	3,156	–0.0833	**<0.001**	**<0.0001**	**<0.0001**
EQR Fish	55	195	–0.0667	0.183	0.1610	0.3957
EQR NTAXA	174	1,241	+0.000	1.0	1.0	0.9232
EQR ASPT	174	1,241	–0.02864	**0.002**	**0.0039**	**0.0046**

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Least duckweed (Lemna minuta)

Sufficient data were available to test all 4 measures of EQR (Figure 4.7). All 3 statistical tests detected a significant difference in the EQR of macrophytes, with the EQR lower in reach years where least duckweed was present (Table 4.7).

It is plausible that thick layers of least duckweed could suppress growth of submerged plants through competition for light. But as least duckweed is included in the taxa considered by the macrophyte tool (River LEAFPACS2) with a RMNI score of 9.21

(WFD UKTAG 2014b), it is not clear whether there is a biological basis to any influence on EQR of macrophytes or not.

Reach year scale analysis: least duckweed (*Lemna minuta*)

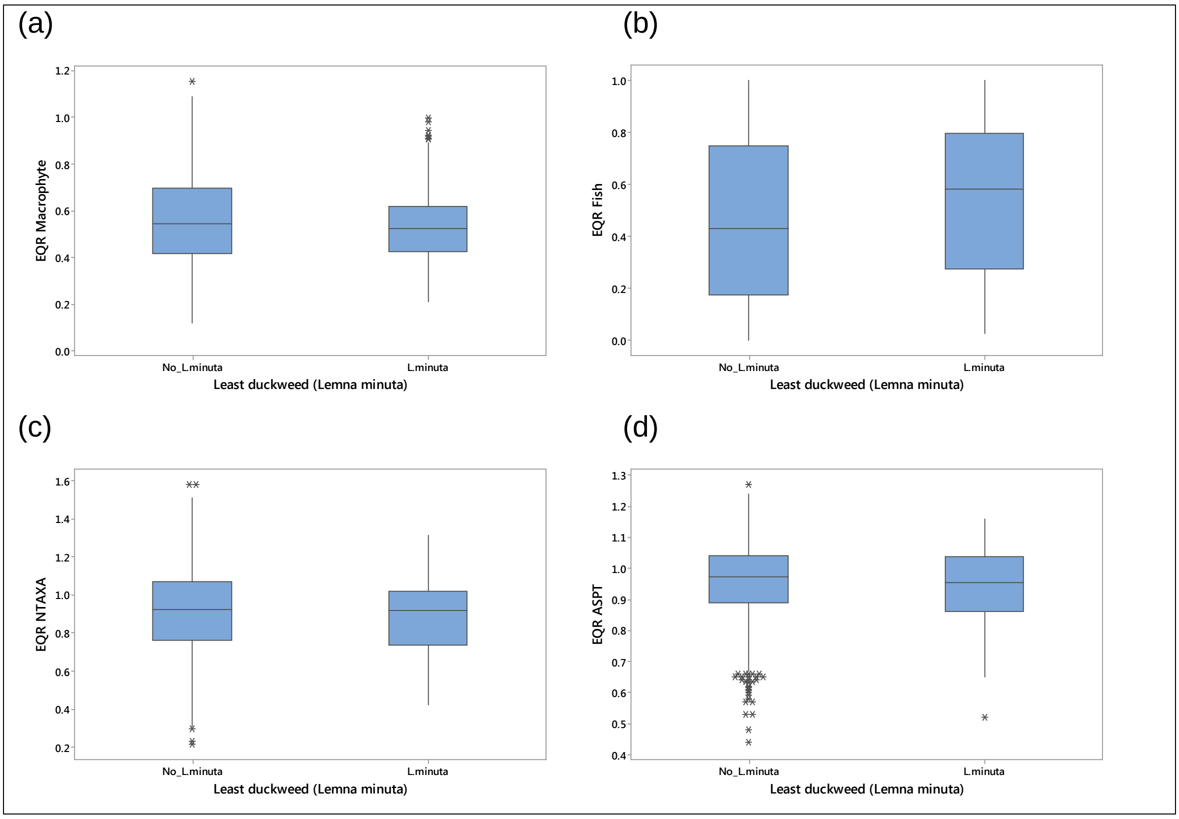


Figure 4.7 Box plots of EQR of (a) macrophytes, (b) fish, and invertebrate (c) NTAXA and (d) ASPT from reach years with and without least duckweed

Table 4.7 Results of statistical tests of the association between least duckweed and EQR at the reach year scale

	N_1	**N_0**	**Difference between means**	***p***		
				t-test	**Logistic**	**Q_{90}**
EQR Macrophyte	415	3,188	–0.04125	**<0.001**	**<0.0001**	**<0.0001**
EQR Fish	32	301	+0.0719	0.244	0.2267	0.3315
EQR NTAXA	84	1,270	–0.0288	0.255	0.2541	0.0124
EQR ASPT	84	1,270	–0.0207	0.135	0.1332	0.5712

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Giant hogweed (Heracleum mantegazzianum)

Sufficient data were available to test all 4 measures of EQR (Figure 4.8). All 3 statistical tests detected significant difference in EQR of fish in reach years where giant hogweed was present (Table 4.8).

The EQR of fish was lower in reach years with giant hogweed (mean difference = –0.1937). The t-test also detected a difference in mean EQR of macrophytes, although the other 2 tests were not significant (logistic regression marginally so).

The biological basis for an influence on fish is not clear and may be a consequence of the lower numbers of reach years used in this analysis (N_1 = 22, N_0 = 202; Table 4.8). This is in turn a consequence of increased sediment inputs as a result of giant hogweed colonisation of riverbanks, or the co-occurrence giant hogweed with other stressors.

Reach year scale analysis: giant hogweed (*Heracleum mantegazzianum*)

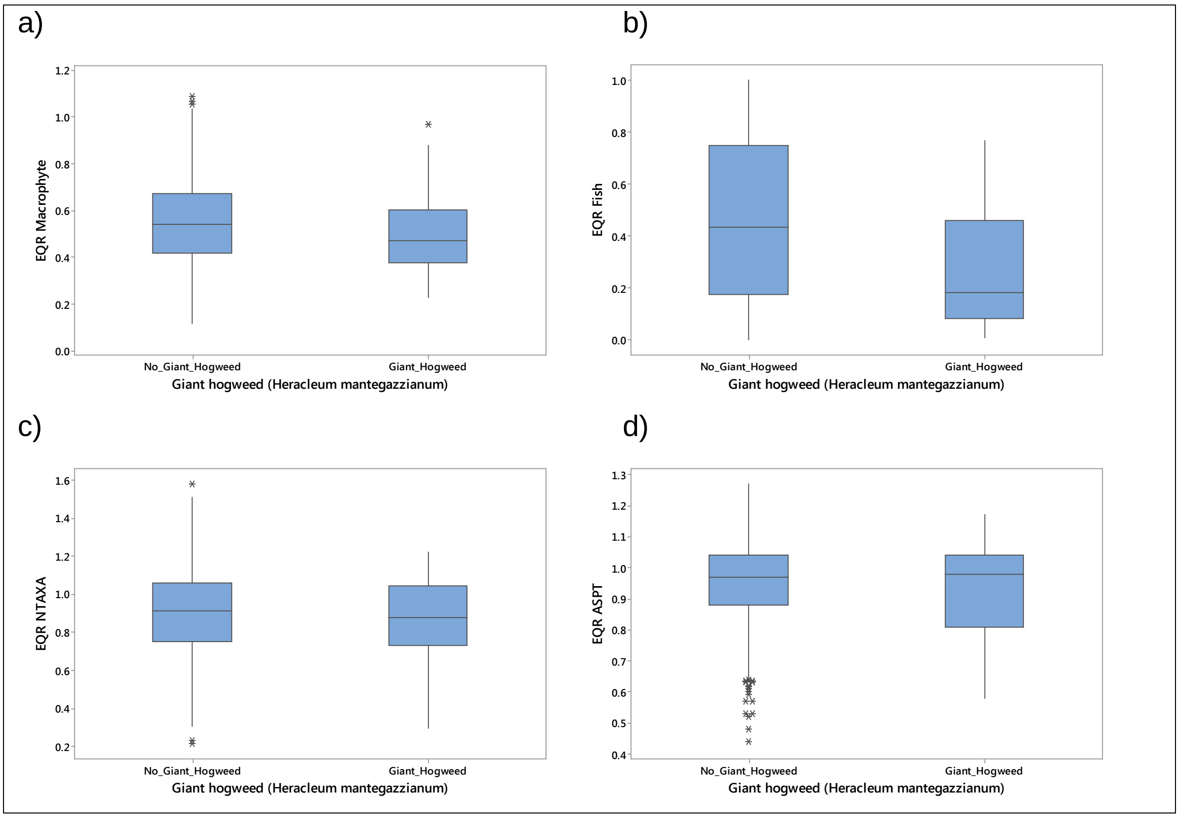


Figure 4.8 Box plots of EQR of (a) macrophytes, (b) fish, and invertebrate (c) NTAXA and (d) ASPT from reach years with and without giant hogweed

Table 4.8 Results of statistical tests of the association between giant hogweed and EQR at the reach year scale

	N_1	N_0	Difference between means	*p*		
				t-test	Logistic	Q_{90}
EQR Macrophyte	40	3,474	–0.0574	**0.038**	0.0545	0.2081
EQR Fish	22	202	–0.1937	**0.002**	**0.0084**	**0.0022**
EQR NTAXA	46	1,315	–0.0468	0.172	0.1682	0.0853
EQR ASPT	46	1,315	–0.0290	0.220	0.1225	0.7063

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Himalayan balsam (Impatiens glandulifera)

Sufficient data were available to test all 4 measures of EQR (Figure 4.9). All 3 statistical tests detected a significant difference in the EQR of macrophytes (Table 4.9), although the mean difference (–0.03339) was small relative to class width (0.2; WFD UKTAG 2014b). Both the t-test and logistic regression detected a significant difference in the EQR of NTAXA and ASPT, but quantile regression did not – suggesting there was less of difference the upper limit of EQR. Again, differences in mean EQR were relatively small (Table 4.9).

Reach year scale analysis: Himalayan balsam (*Impatiens glandulifera*)

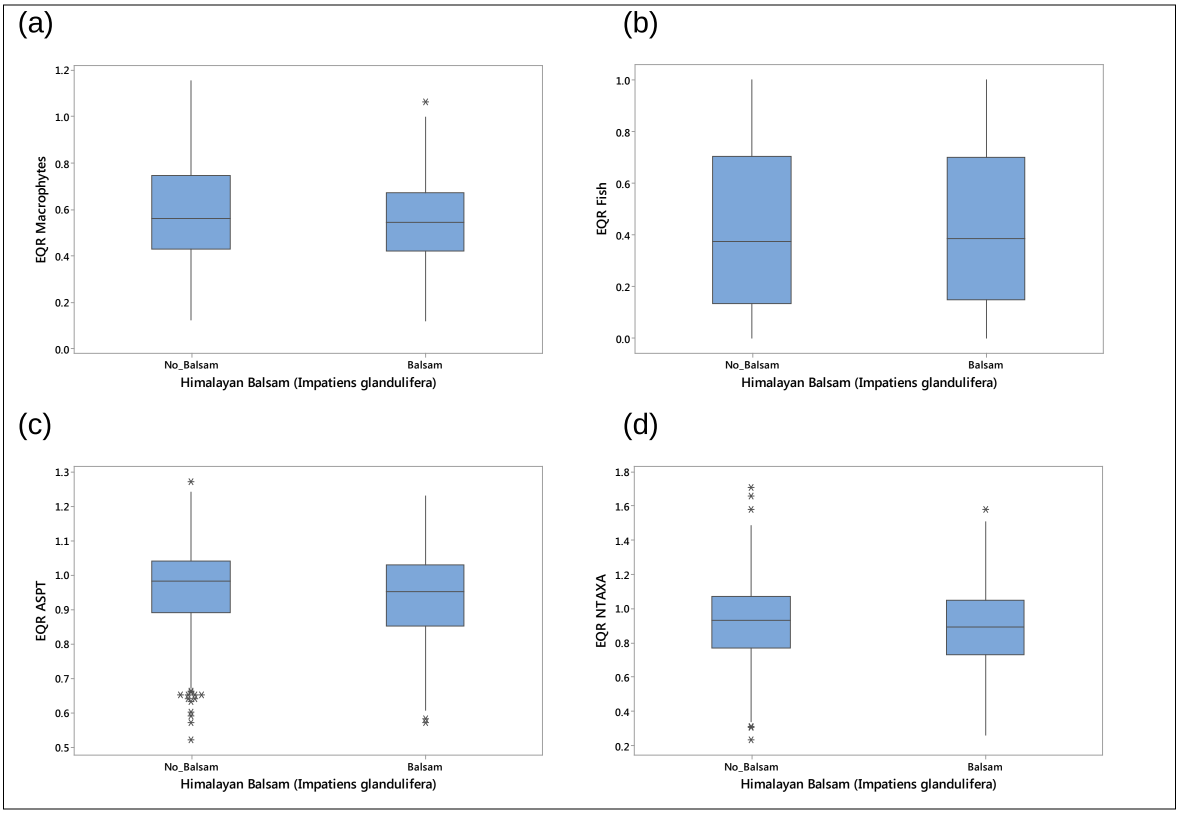


Figure 4.9 Box plots of EQR of (a) macrophytes, (b) fish, and invertebrate (c) NTAXA and (d) ASPT from reach years with and without Himalayan balsam

Table 4.9 Results of statistical tests of the association between Himalayan balsam and EQR at the reach year scale

	N_1	**N_0**	**Difference between means**	***p***		
				t-test	**Logistic**	**Q_{90}**
EQR Macrophyte	787	2,294	–0.03339	**<0.001**	**<0.0001**	**<0.0001**
EQR Fish	184	144	–0.0003	0.994	0.9842	0.4514
EQR NTAXA	479	915	–0.0327	**0.011**	**0.0102**	**0.2617**
EQR ASPT	479	915	–0.02807	**<0.001**	**<0.0001**	**0.4053**

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Zander (Sander lucioperca)

Sufficient data were available to test measures of EQR for fish and invertebrates; however, there were insufficient data to test association with the EQR of macrophytes (Figure 4.10). All 3 statistical tests detected a significant difference in the EQR of fish (Table 4.10). Reach years where zander were present had a substantially higher EQR of fish (+0.3479).

It is plausible that the presence of zander makes fish more catchable by altering the size structure of populations, but it is also possible that zander prefer sites with good fish populations.

Reach year scale analysis: zander (*Sander lucioperca*)

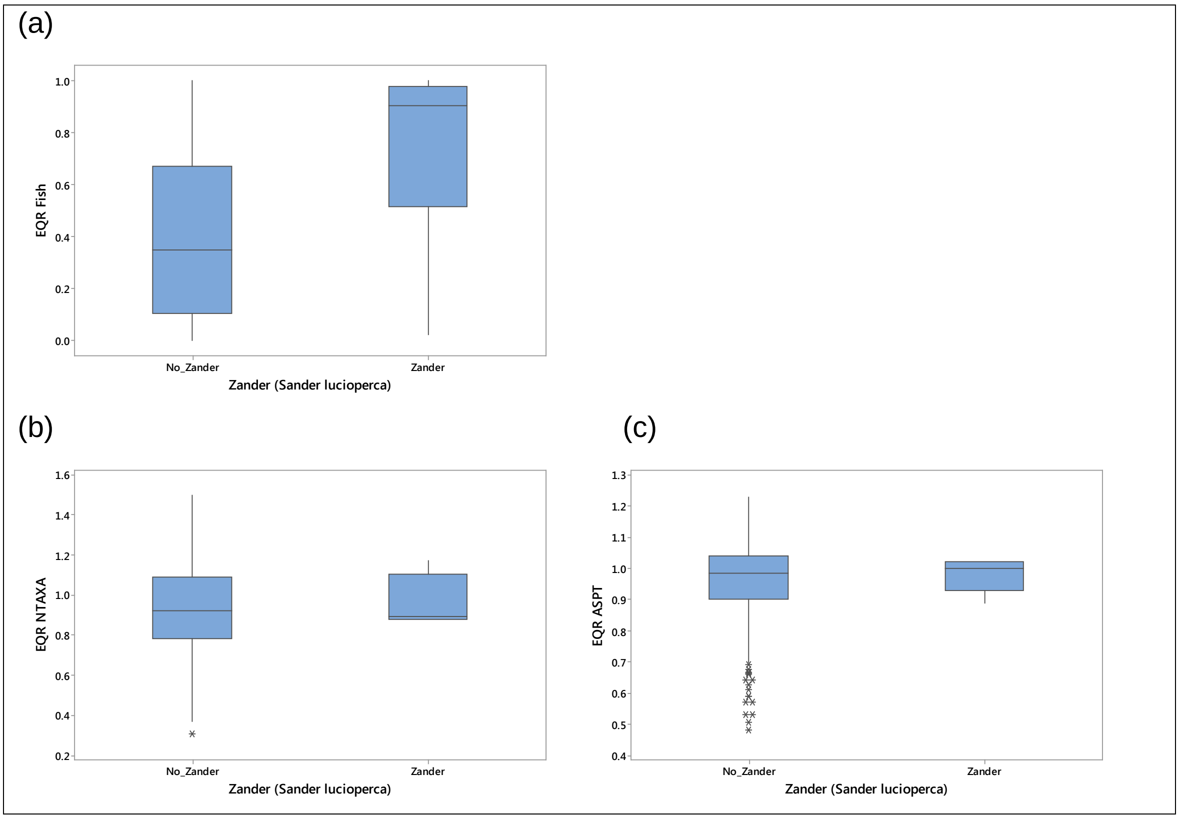


Figure 4.10 Box plots of EQR of (a) fish, and invertebrate (b) NTAXA and (c) ASPT from reach years with and without zander

Table 4.10 Results of statistical tests of the association between zander and EQR at the reach year scale

	N_1	**N_0**	**Difference between means**	***p***		
				t-test	**Logistic**	**Q_{90}**
EQR Macrophyte	1					
EQR Fish	44	5,159	+0.3479	**<0.001**	**<0.0001**	**0.0413**
EQR NTAXA	5	425	+0.0513	0.432	0.5943	0.8993
EQR ASPT	5	425	+0.0185	0.500	0.7423	0.6269

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

Sunbleak (Leucaspius delineatus)

Although sufficient data were available to test all 4 measures of EQR (Figure 4.11), the number of reach years with sunbleak was relatively low. The t-test and logistic regression detected a difference in the EQR of fish where reach years with sunbleak returned a substantially lower EQR, but this difference was not detected by quantile regression (Table 4.11). The low numbers of reach years used in the analyses may have had an influence on these findings.

Reach year scale analysis: sunbleak (*Leucaspius delineatus*)

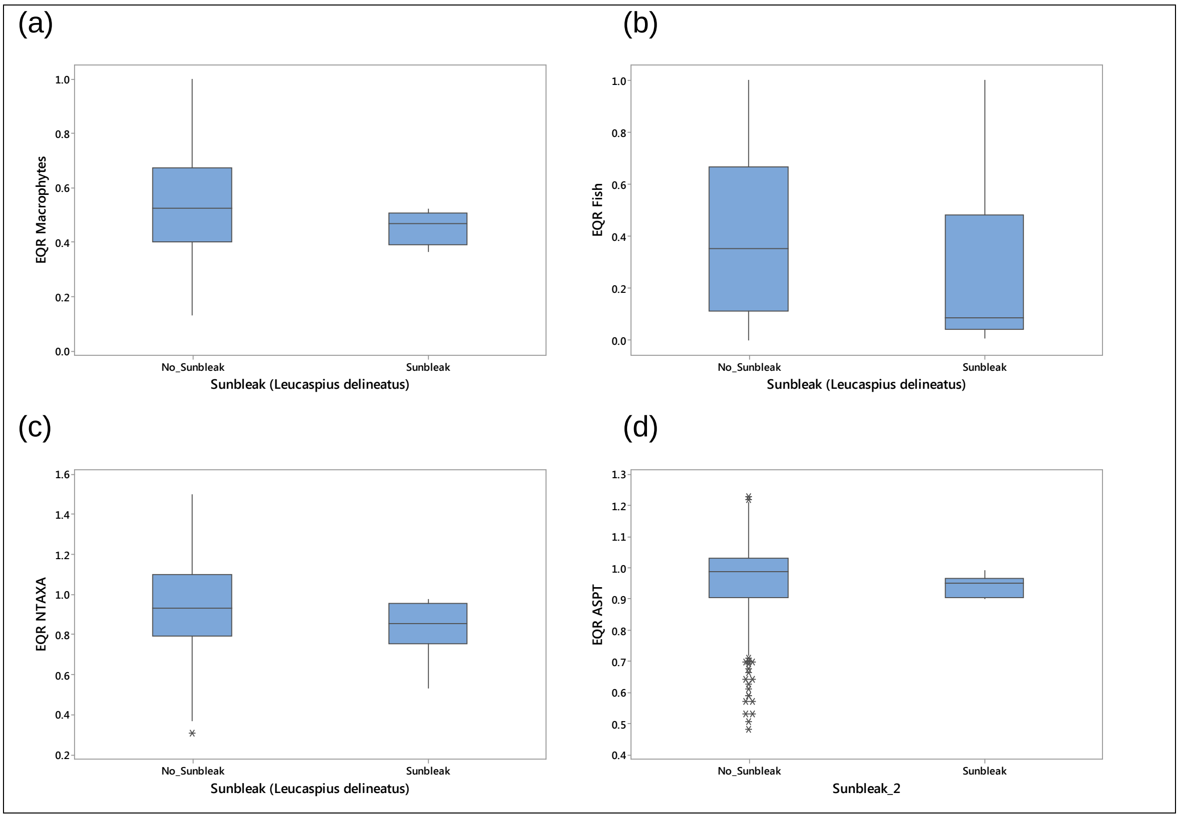


Figure 4.11 Box plots of EQR of (a) macrophytes, (b) fish, and invertebrate (c) NTAXA and (d) ASPT from reach years with and without sunbleak

Table 4.11 Results of statistical tests of the association between sunbleak and EQR at the reach year scale

	N_1	**N_0**	**Difference between means**	***p***		
				t-test	**Logistic**	**Q_{90}**
EQR Macrophyte	4	186	–0.1001	0.051	0.3371	0.2396
EQR Fish	22	5,357	–0.1522	**0.035**	**0.0283**	0.3166
EQR NTAXA	6	466	–0.0992	0.193	0.2518	0.3085
EQR ASPT	6	466	–0.0223	0.188	0.6521	0.4986

Notes: N_1 = number of reach years where the INNS was present.
N_0 = number of reach years where the INNS was absent.
Statistically significant results after shown in bold.

4.4.2 Water body scale over WFD reporting periods

Data were extracted at the water body reporting period scale and statistical tests of their impact on EQR carried out for the following INNS:

- zebra mussel (*Dreissena polymorpha*), MV
- signal crayfish (*Pacifastacus leniusculus*), MR
- floating pennywort (*Hydrocotyle ranunculoides*), MR
- common carp (*Cyprinus carpio*), MR
- demon shrimp (*Dikerogammarus haemobaphes*), MO
- Nuttall's pondweed (*Elodea nuttallii*), MO
- Himalayan balsam(*Impatiens glandulifera*), MO
- zander (*Sander lucioperca*), MO
- sunbleak (*Leucaspius delineatus*), MN

The choice of INNS to be considered was influenced by the occurrence of INNS in terms of the number of water bodies where they were found and their spread during the period for which EQR data were available. For the BACI approach to work, the INNS under consideration had to be recorded in the impacted water bodies part way through the time series of EQR data. Insufficient data to conduct the test were available for those INNS that were already well-established before the period of EQR data and those INNS that had only recently arrived.

Statistically significant effects were detected for several of the INNS considered (Tables 4.12 to 4.20, Figures 4.12 to 4.19). However, the majority of the significant results were for main effects:

- a change over time resulting in a difference between the before and after periods (B/A) which affected both the control and impacted groups
- an inherent difference between the water bodies placed between the control and impacted groups (INNS under consideration) irrespective of whether the INNS was present

These main effects provide no information on the impact of the INNS.

Overall, the EQR of ASPT (Tables 4.12, 4.13, 4.14, 4.16; Figures 4.12, 4.13, 4.15, 4.17), NTAXA (Tables 4.12 to 4.14; Figures 4.12 to 4.15, 4.18) and fish (Tables 4.14, 4.15; Figures 4.15, 4.16) tended to increase with time (indicated by a significant B/A main effect), indicating a general improvement in the status of these BQEs.

A significant difference between the control and impacted groups of water bodies was detected for the EQR of ASPT and NTAXA in the test with floating pennywort (Table 4.14; Figure 4.15) and NTAXA in the test with demon shrimp (Table 4.16; Figure 4.17), suggesting differences in the condition of the sites allocated to the 2 groups that are not related to the presence of the INNS.

Nevertheless, a significant interaction (B/A * INNS) was detected indicating a positive impact of signal crayfish (Table 4.13b; Figure 4.13c) and Himalayan balsam (Table 4.18; Figure 4.19) on the EQR of ASPT, and negative impacts of signal crayfish (Table 4.13a; Figure 4.13a, 4.14) and demon shrimp (Table 4.16a; Figures 4.17a, 4.18) on EQR NTAXA.

Zebra mussel (Dreissena polymorpha)

Data were available to test the effect of invasion of water bodies by zebra mussel in all 3 time periods for all measures of EQR except macrophytes (Figure 4.12, Table 4.12). But due to the later introduction of the LEAFPACS tool, ‘before’ EQR data were lacking for water bodies invaded in the early WFD reporting period, an issue common to all the species tested.

Water body reporting period scale analysis: zebra mussel (*Dreissena polymorpha*)

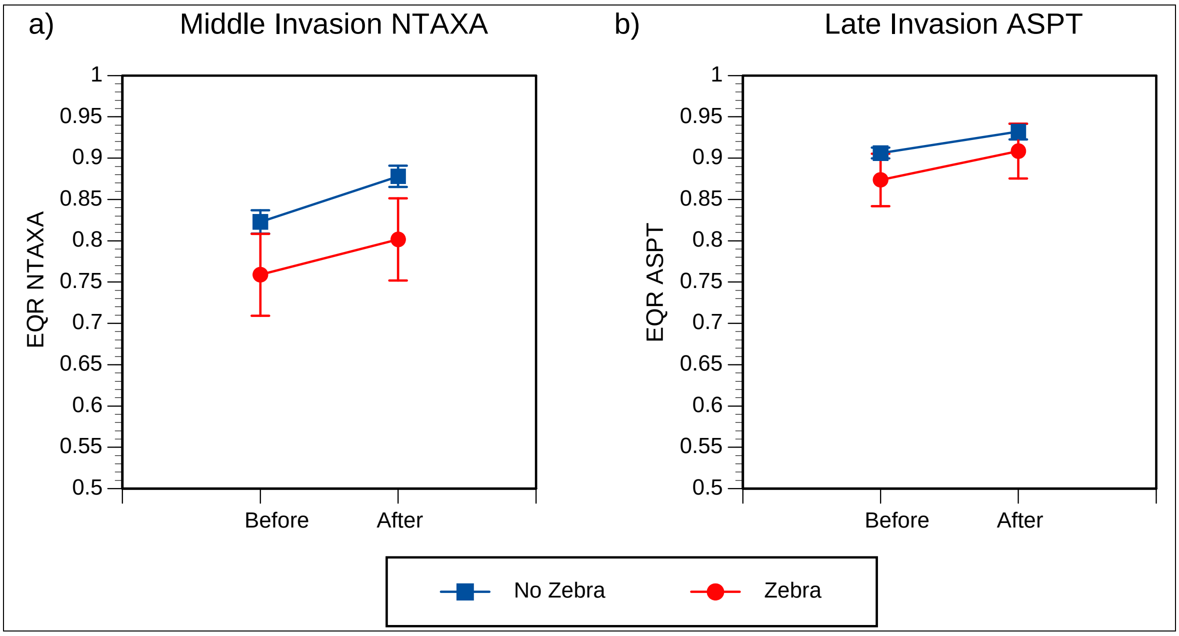


Figure 4.12 Interaction plot showing significant results of BACI test of impact of zebra mussel on WFD measures of EQR

Table 4.12 Numbers of water bodies (samples) used and results of asymmetrical analysis of variance (ANOVA) ‘BACI’ test of impact of zebra mussel on EQR

(a) Early occurrence of zebra mussel in water body

	N_i^1		N_c^1		***p***		
					B/A	**Zebra**	**B/A * Zebra**
EQR Macrophyte							
EQR Fish	5	(65)	89	(1,216)	0.9298	0.9911	0.7004
EQR NTAXA	12	(88)	296	(2,670)	0.2707	0.4165	0.2034
EQR ASPT	12	(88)	296	(2,670)	0.1504	0.5251	0.9748

(b) Middle occurrence of zebra mussel in water body

	N_i^1		N_c^1		***p***		
					B/A	**Zebra**	**B/A * Zebra**
EQR Macrophyte	4	(18)	69	(596)	0.8760	0.8050	0.5579
EQR Fish	11	(79)	89	(1,216)	0.4676	0.2197	0.7683
EQR NTAXA	14	(100)	549	(4,489)	**0.0233**	0.1329	0.6665
EQR ASPT	14	(100)	549	(4,489)	0.0877	0.0690	0.1886

(c) Late occurrence of zebra mussel in water body

	N_i^1		N_c^1		***p***		
					B/A	**Zebra**	**B/A * Zebra**
EQR Macrophyte	6	(18)	64	(568)	0.1766	0.1821	0.5015
EQR Fish	6	(47)	80	(1,133)	0.4432	0.4629	0.5177
EQR NTAXA	14	(75)	517	(4,267)	0.0612	0.5296	0.8500
EQR ASPT	14	(75)	517	(4,267)	**0.0095**	0.3726	0.5626

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

Signal crayfish (Pacifastacus leniusculus)

A significant interaction (B/A * INNS) was detected indicating a positive impact of signal crayfish on EQR of ASPT (Figure 4.13c) and a negative impact on EQR of NTAXA (Figure 4.13a). The difference in mean EQR of NTAXA between the impacted and control group in the after period was substantial (0.104) compared with the class width for this measure of ecological quality (NTAXA class width = 0.2–0.14), equivalent to approximately half to three-quarters of a WFD class.

Although the EQR of ASPT increased with time, it increased more rapidly in water bodies where signal crayfish were detected during the middle WFD reporting period than in the control group. The number of water bodies used in this test (N_i = 28, N_c = 940; Table 4.13b) provide confidence that this difference was due to the presence of signal crayfish. For those water bodies where signal crayfish occurred early, the EQR of NTAXA went down after signal crayfish had been detected, whereas there was no significant change for the control group (Figure 4.13a).

Significant interaction effects were not apparent for all periods of invasion (early, middle and late). This is a consequence of the asymmetric structure of the BACI analysis, where 2 factors influence the probability of detecting an impact of INNS.

The first factor of effect size – the larger the impact of the INNS relative to background variation, the more likely that an effect will be detected. As INNS represent a press disturbance (that is, a sustained impact, likely to become larger over time, as the INNS population becomes established and expands) rather than a pulse disturbance (that is, a temporary, sudden shock, with potential for recovery thereafter), effect size is likely to be associated with the time after invasion. It is clear from the trajectory of change in EQR of NTAXA over time in water bodies where signal crayfish first occurred in the early reporting period (Figure 4.14) that the difference between the impacted and control water bodies became more pronounced with time, increasing the probability of detecting a significant effect where the test includes those water bodies where the INNS has been established longest.

Similarly, the interaction between time and signal crayfish (B/A * Signal) was significant for the EQR of ASPT in water bodies where signal crayfish first occurred in the middle reporting period (Table 4.13b), but not for water bodies where signal crayfish first occurred in the late reporting period (Table 4.13c). This was despite the EQR of ASPT following similar trajectories with time in both (Figure 4.13c, 4.13d). Again, this suggests that the duration of colonisation has an important influence on effect size.

The second factor is replication – the higher the number of replicate measures used to establish mean values, the more likely that an effect will be detected. Here replication comprises both the number of water bodies in the control and impacted groups, and the duration of the before and after period. A longer duration ‘before’ and ‘after’ increases the number of samples used to derive mean values. As a consequence those water bodies where the INNS first occurred in the middle reporting period were more likely to return a significant result, as shown for the impact of signal crayfish on the EQR of ASPT.

The changes in EQR of ASPT and NTAXA are consistent with the findings at the reach year scale, adding further confidence to the findings.

Water body reporting period scale analysis: signal crayfish (*Pacifastacus leniusculus*)

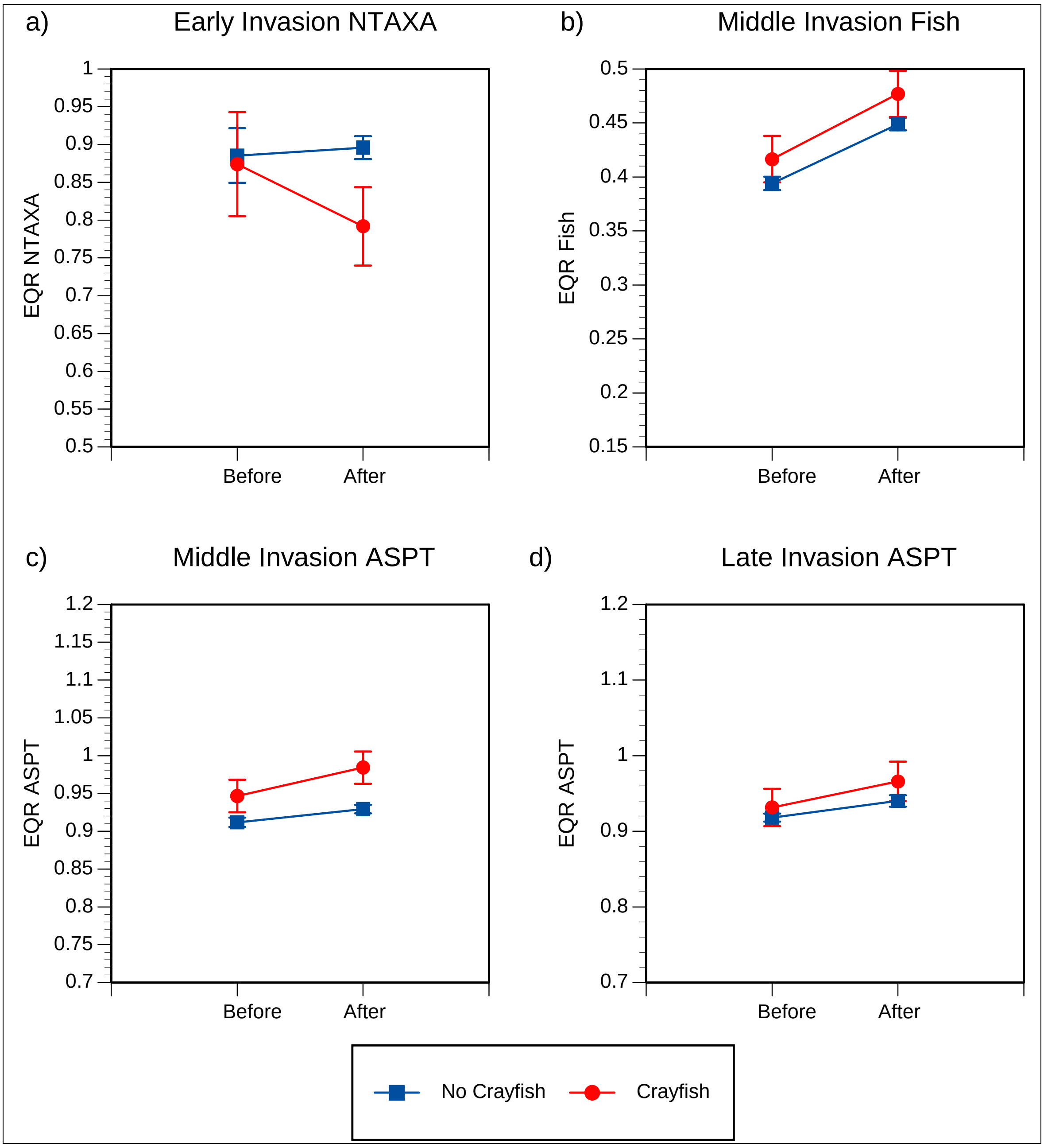


Figure 4.13 Interaction plot showing significant results of BACI test of impact of signal crayfish on WFD measures of EQR

Table 4.13 Numbers of water bodies (samples) used and results of asymmetrical ANOVA 'BACI' test of impact of signal crayfish (*Pacifastacus leniusculus*) on EQR

(a) Early occurrence of signal crayfish in water body

	N_i[1]		N_c[1]		*p*		
					B/A	**Crayfish**	**B/A * Crayfish**
EQR Macrophyte							
EQR Fish	9	(72)	260	(2,535)	0.5523	0.9720	0.3575
EQR NTAXA	13	(100)	431	(3,398)	0.3978	0.2663	**0.0252**
EQR ASPT	13	(100)	431	(3,398)	0.0883	0.5991	0.8762

(b) Middle occurrence of signal crayfish in water body

	N_i[1]		N_c[1]		*p*		
					B/A	**Crayfish**	**B/A * Crayfish**
EQR Macrophyte	6	(43)	59	(335)	0.7011	0.4549	0.6059
EQR Fish	22	(171)	260	(2,535)	**0.0138**	0.5764	0.9054
EQR NTAXA	28	(247)	940	(6,212)	0.1052	0.3203	0.8200
EQR ASPT	28	(247)	940	(6,212)	**0.0003**	0.0318	**0.0090**

(c) Late occurrence of signal crayfish in water body

	N_i[1]		N_c[1]		*p*		
					B/A	**Crayfish**	**B/A * Crayfish**
EQR Macrophyte							
EQR Fish	10	(78)	260	(2,535)	0.5216	0.2719	0.8817
EQR NTAXA	21	(131)	810	(5,550)	0.3684	0.6168	0.0301
EQR ASPT	21	(131)	810	(5,550)	**0.0023**	0.4215	0.2943

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

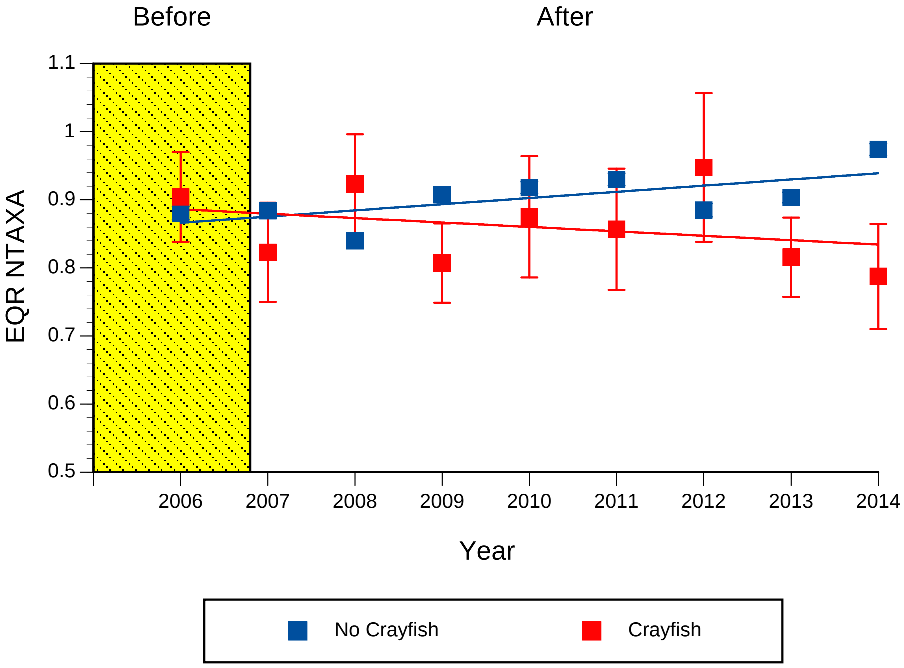


Figure 4.14 Variation in mean EQR of NTAXA (± standard error) with time for water bodies where signal crayfish first occurred in the early reporting period compared with control water bodies

*Floating pennywort (*Hydrocotyle ranunculoides*)*

Data were available to test the effect of invasion of water bodies by floating pennywort in all 3 time periods for all measures of EQR except macrophytes, for which data were for the early and late reporting periods (Table 4.14). The main effects of time (B/A) were apparent for the EQR of NTAXA, ASPT and fish, and of floating pennywort for the EQR of NTAXA and ASPT. The latter result casts some doubt on the association between floating pennywort and the EQR of ASPT found at the reach year scale (Table 4.3; Figure 4.3). Sites allocated to the impact group had a significantly lower EQR of ASPT irrespective of whether floating pennywort was there or not (Figure 4.15).

Although field teams have reported that floating pennywort can reduce the efficiency of fishing, no effect of the invasion of water bodies by floating pennywort on EQR of fish was detected.

Water body reporting period scale analysis: floating pennywort (*Hydrocotyle ranunculoides*)

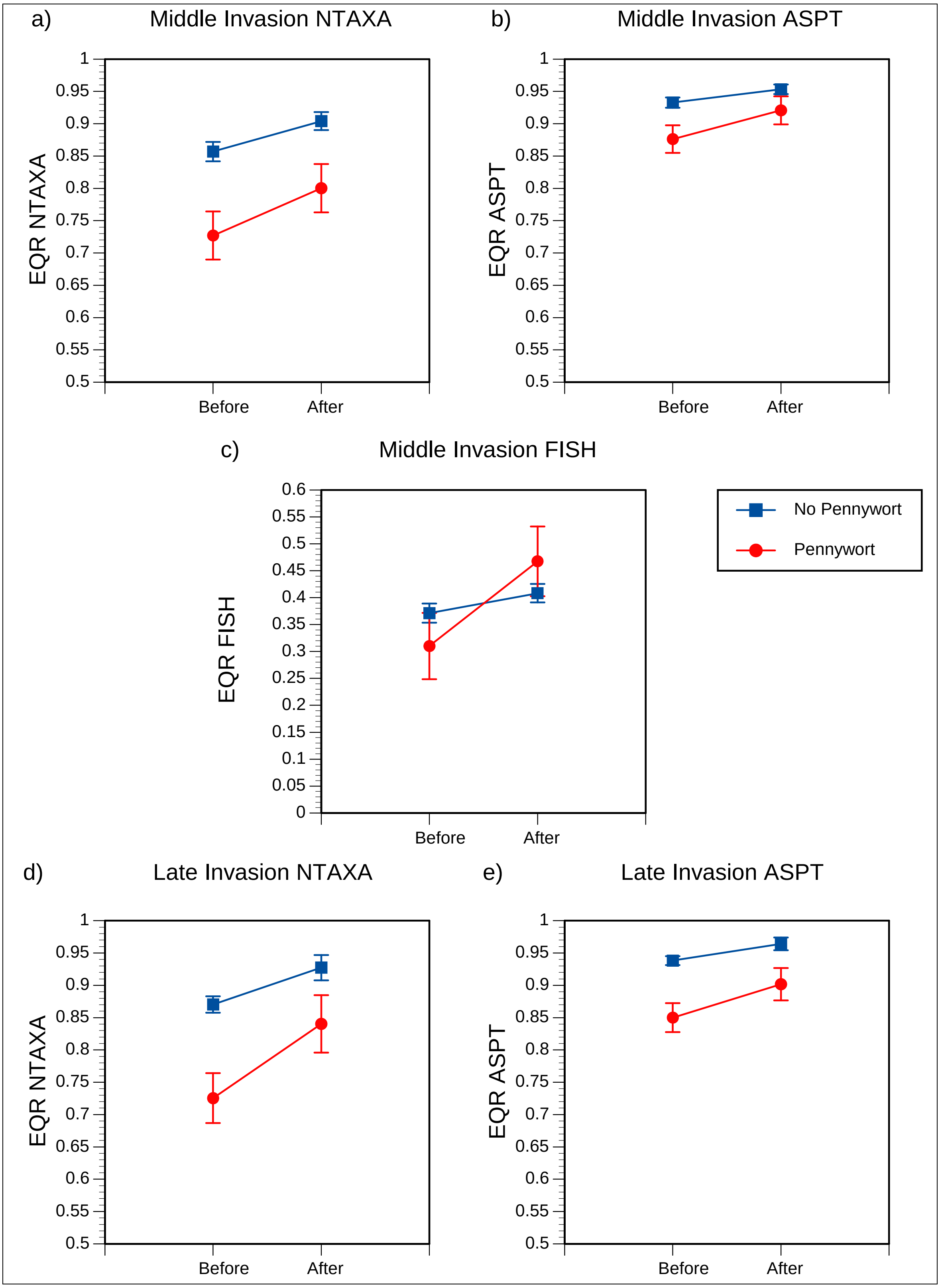


Figure 4.15 Interaction plot showing significant results of BACI test of impact of floating pennywort on WFD measures of EQR

Table 4.14 Numbers of water bodies (samples) used and results of asymmetrical ANOVA 'BACI' test of impact of floating pennywort on EQR

(a) Early occurrence of floating pennywort in water body

	N_i^1		**N_c^1**		***p***		
					B/A	**Pennywort**	**B/A * Pennywort**
EQR Macrophyte							
EQR Fish	8	(90)	164	(2,053)	0.7645	0.3736	0.6083
EQR NTAXA	4	(66)	266	(2,388)	0.3363	0.5174	0.6008
EQR ASPT	4	(66)	266	(2,388)	0.0284	0.3880	0.2398

(b) Middle occurrence of floating pennywort in water body

	N_i^1		**N_c^1**		***p***		
					B/A	**Pennywort**	**B/A * Pennywort**
EQR Macrophyte	8	(33)	30	(195)	0.2890	0.7273	0.7937
EQR Fish	13	(169)	164	(2,053)	**0.0025**	0.9879	0.0451
EQR NTAXA	18	(160)	267	(2,398)	**0.0088**	**0.0009**	0.2729
EQR ASPT	18	(160)	267	(2,398)	**0.0048**	0.0320	0.0447

(c) Late occurrence of floating pennywort in water body

	N_i^1		**N_c^1**		***p***		
					B/A	**Pennywort**	**B/A * Pennywort**
EQR Macrophyte							
EQR Fish	9	(82)	164	(2,053)	0.3135	0.7247	0.6616
EQR NTAXA	17	(143)	266	(2,388)	**0.0028**	**0.0023**	0.0609
EQR ASPT	17	(143)	266	(2,388)	**0.0047**	**0.0008**	0.0896

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

*Common carp (*Cyprinus carpio*)*

There were only sufficient data to test the effect of invasion of water bodies by common carp for measures of EQR of fish, although this was possible for all 3 time periods. A significant main effect of time (B/A) was detected on the EQR of fish for the middle time period (Table 4.15b; Figure 4.16).

Water body reporting period scale analysis: common carp (*Cyprinus carpio*)

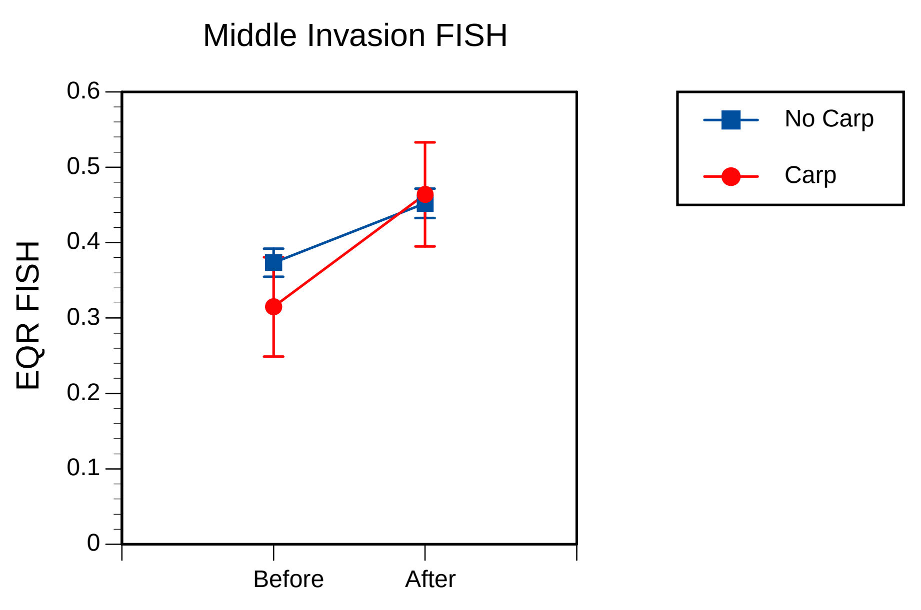


Figure 4.16 Interaction plot showing significant results of BACI test of impact of common carp on WFD measures of EQR

Table 4.15 Numbers of water bodies (samples) used and results of asymmetrical ANOVA 'BACI' test of impact of common carp on EQR

(a) Early occurrence of common carp in water body

	N_i[1]		N_c[1]		*p*		
					B/A	**Carp**	**B/A * Carp**
EQR Macrophyte							
EQR Fish	24	(160)	161	(1,675)	0.8311	0.7433	0.1203
EQR NTAXA							
EQR ASPT							

(b) Middle occurrence of common carp in water body

	N_i[1]		N_c[1]		*p*		
					B/A	**Carp**	**B/A * Carp**
EQR Macrophyte							
EQR Fish	11	(73)	161	(1,675)	**0.0017**	0.6999	0.2780
EQR NTAXA							
EQR ASPT							

(c) Late occurrence of common carp in water body

	N_i[1]		**N_c[1]**		***p***		
					B/A	**Carp**	**B/A * Carp**
EQR Macrophyte							
EQR Fish	7	(48)	161	(1,675)	0.9734	0.7745	0.4923
EQR NTAXA							
EQR ASPT							

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

Demon shrimp (Dikerogammarus haemobaphes)

For demon shrimp, the significant interaction (B/A * Demon; Table 4.16a) reflected no change in the EQR of NTAXA in the impacted group of water bodies between the before and after time periods, relative to an increase in the control group (Figure 4.17, 4.18).

As the general trend in the EQR of NTAXA across all the other datasets was to increase with time, this is a substantial result. The presence of demon shrimp constrains the recovery in the EQR of NTAXA that was apparent elsewhere, with the difference approximately 0.1 EQR by the end of the time series (Figure 4.18). This difference equates to approximately half to three-quarters of a WFD class.

Due to the later arrival of demon shrimp than signal crayfish, there were insufficient data to test the effect of the first occurrence of demon shrimp in the early reporting period, but it is likely that impacts will become more pronounced with the duration of invasion.

Main effects of time (B/A) were detected for the EQR of ASPT (Table 4.16; Figure 4.17c, 4.17d) and for the EQR of NTAXA for water bodies invaded in the late period (Table 4.16b; Figure 4.17b).

Water body reporting period scale analysis: demon shrimp (*Dikerogammarus haemobaphes*)

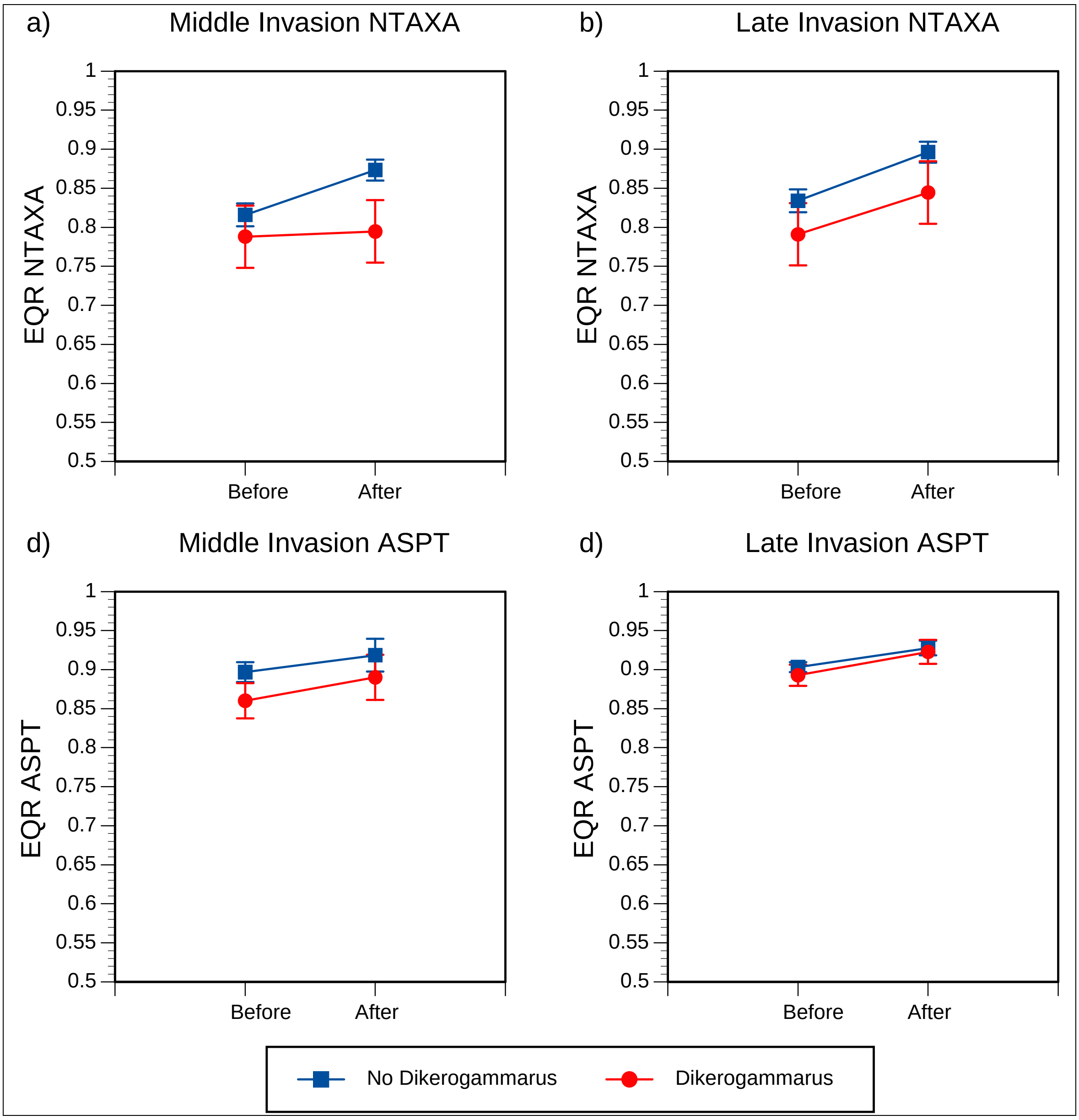


Figure 4.17 Interaction plot showing significant results of BACI test of impact of demon shrimp on WFD measures of EQR

Table 4.16 Numbers of water bodies (samples) used and results of asymmetrical ANOVA 'BACI' test of impact of demon shrimp on EQR

(a) Middle occurrence of demon shrimp in water body

	N_i^1		N_c^1		*p*		
					B/A	**Demon**	**B/A * Demon**
EQR Macrophyte	16	(86)	79	(471)	0.2259	0.8051	0.0863
EQR Fish	26	(126)	255	(2,852)	0.5353	0.7172	0.3665
EQR NTAXA	19	(198)	535	(4,425)	0.1232	0.1577	**0.0084**
EQR ASPT	19	(198)	535	(4,425)	**0.0140**	0.1951	0.3097

(b) Late occurrence of demon shrimp in water body

	N_i[1]		N_c[1]		***p***		
					B/A	**Demon**	**B/A * Demon**
EQR Macrophyte	26	(108)	79	(471)	0.0943	0.2297	0.8217
EQR Fish	36	(477)	255	(2,852)	0.5697	0.1236	0.8936
EQR NTAXA	70	(660)	535	(4,425)	0.0374	**0.0250**	0.5114
EQR ASPT	70	(660)	535	(4,425)	**0.0215**	0.5754	0.3243

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

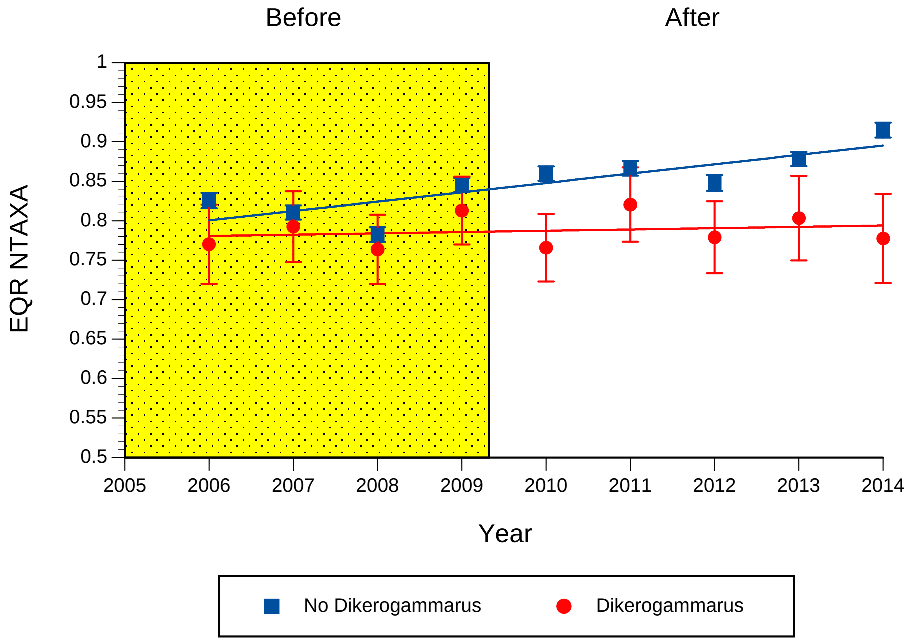


Figure 4.18 Variation in mean EQR of NTAXA (± standard error) with time for water bodies where demon shrimp first occurred in the early reporting period compared with control water bodies

Nuttall's pondweed (Elodea nuttallii)

There were sufficient data to test the effect of invasion of water bodies by Nuttall's pondweed in all 3 periods for measures of the EQR of invertebrates, for fish in the early and middle periods and macrophytes in the middle period. No statistically significant effects were detected.

Table 4.17 Numbers of water bodies (samples) used and results of asymmetrical ANOVA 'BACI' test of impact of Nuttall's pondweed on EQR

(a) Early occurrence of Nuttall's pondweed in water body

	N_i[1]		**N_c[1]**		***p***		
					B/A	**Elodea**	**B/A * Elodea**
EQR Macrophyte							
EQR Fish	5	(35)	86	(723)	0.4465	0.5683	0.3088
EQR NTAXA	7	(60)	78	(634)	0.5890	0.8972	0.1435
EQR ASPT	7	(60)	78	(634)	0.3229	0.7264	0.9211

(b) Middle occurrence of Nuttall's pondweed in water body

	N_i[1]		**N_c[1]**		***p***		
					B/A	**Elodea**	**B/A * Elodea**
EQR Macrophyte	2	(12)	25	(138)	0.6051	0.9258	0.2685
EQR Fish	14	(96)	86	(723)	0.8476	0.7597	0.6017
EQR NTAXA	15	(84)	177	(1,316)	0.2425	0.8510	0.8374
EQR ASPT	15	(84)	177	(1,316)	0.1250	0.7982	0.8682

(c) Late occurrence of Nuttall's pondweed in water body

	N_i[1]		**N_c[1]**		***p***		
					B/A	**Elodea**	**B/A * Elodea**
EQR Macrophyte							
EQR Fish							
EQR NTAXA	11	(42)	173	(1,289)	0.4618	0.5363	0.1795
EQR ASPT	11	(42)	173	(1,289)	0.1866	0.0615	0.4513

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

Himalayan balsam (Impatiens glandulifera)

Here the significant interaction reflected a decline in the EQR of ASPT in the control group of water bodies over time relative to no change in the impacted group (Figure 4.20). As the general trend in the EQR of ASPT across all other datasets was to increase with time, this result is likely to be a consequence of the relatively low number of water bodies in the control group rather than a substantial influence of Himalayan balsam on the invertebrate community of invaded sites.

Himalayan balsam is widespread in England, with records from almost every 10km grid square (Alpha Hull Area = 129,297km^2: Figure B.42 in Appendix B). Compared with other INNS, the number of water bodies that did not have any records of the species was low (N_c = 19) relative to those where it occurred (N_i = 191). There is the possibility that the absence of Himalayan balsam coincided with some other factor influencing the EQR of ASPT in these water bodies.

Water bodyreporting period scale analysis: Himalayan balsam (*Impatiens glandulifera*)

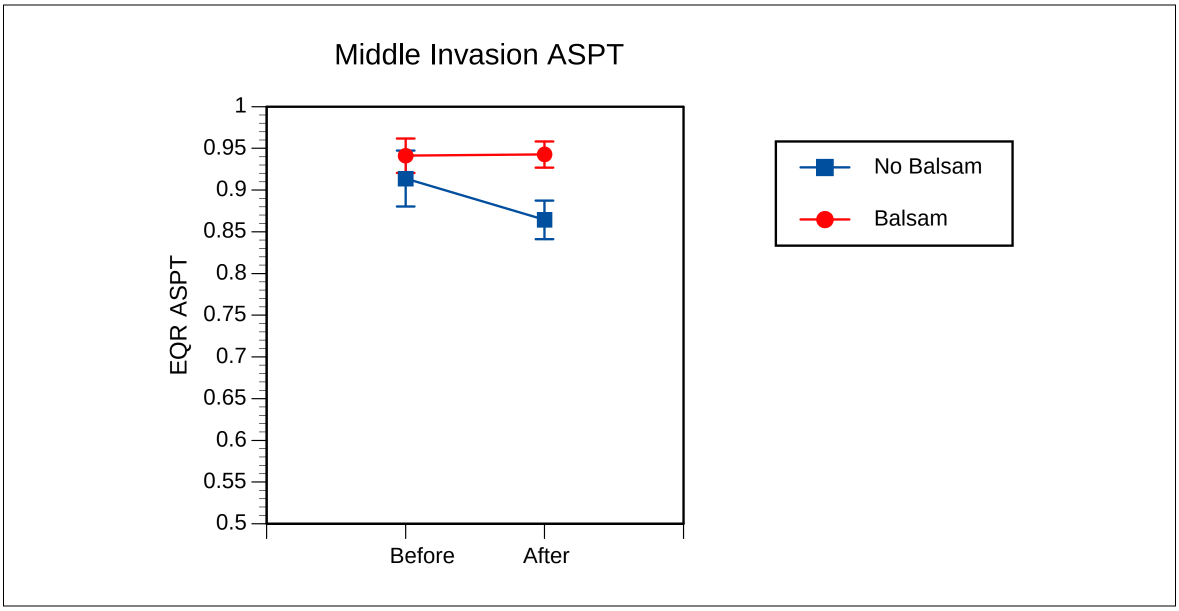


Figure 4.19 Interaction plot showing significant results of BACI test of impact of Himalayan balsam on WFD measures of EQR

Table 4.18 Numbers of water bodies (samples) used and results of asymmetrical ANOVA 'BACI' test of impact of Himalayan balsam on EQR

Middle occurrence of Himalayan balsam in water body

	N_i[1]		N_c[1]		*p*		
					B/A	**Balsam**	**B/A * Balsam**
EQR Macrophyte	4	(14)	11	(26)	0.9763	0.2498	0.9460
EQR Fish	23	(121)	9	(34)	0.3000	0.4995	0.6486
EQR NTAXA	44	(191)	19	(75)	0.1195	0.0717	0.2961
EQR ASPT	44	(191)	19	(75)	0.2389	0.0645	**0.0396**

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

Zander (Sander lucioperca*)*

There were only sufficient data to test the effect of invasion of water bodies by zander in all the middle period for measures of EQR of for fish. No statistically significant effects were detected (Table 4.19).

Water body reporting period scale analysis: zander *(Sander lucioperca*)

Table 4.19 Numbers of water bodies (samples) used and results of asymmetrical ANOVA 'BACI' test of impact of zander on EQR

Middle occurrence of zander in water body

	N_i[1]		**N_c[1]**		***p***		
					B/A	**Zander**	**B/A * Zander**
EQR Macrophyte							
EQR Fish	4	(33)	29	(253)	0.2200	0.8772	0.0712
EQR NTAXA							
EQR ASPT							

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

Sunbleak (Leucaspius delineatus)

There were only sufficient data to test the effect of invasion of water bodies by sunbleak in all the late period for measures of EQR of for fish. No statistically significant effects were detected.

Water body reporting period scale analysis: sunbleak *(Leucaspius delineatus*)

Table 4.20 Numbers of water bodies (samples) used and results of asymmetrical ANOVA 'BACI' test of impact of sunbleak on EQR

Late occurrence of sunbleak in water body

	N_i[1]		**N_c[1]**		***p***		
					B/A	**Sunbleak**	**B/A * Sunbleak**
EQR Macrophyte							
EQR Fish	4	(57)	54	(748)	0.4119	0.9204	0.6290
EQR NTAXA							
EQR ASPT							

Notes: [1] N_i = number of impacted water bodies used in the analysis with the number of EQR values shown in brackets. N_c = corresponding values for control

Statistically significant results after shown in bold. Where significant, the interaction between B/A * INNS indicates an impact.

4.5 Discussion

Attributing any difference in measures of ecological quality to INNS through later data analysis is difficult. Conducting the analysis at 2 different scales increased the probability of detecting differences in measures of ecological quality and attributing any impact to invasive species. The strongest evidence is obtained where the results from both scales concur. Furthermore, by robust replication and including comparable data from control sites, the ability to attribute any differences detected in measures of ecological quality to INNS is greatly enhanced.

A detailed analysis of the impact of INNS on ecological quality as measured by the WFD tools LEAFPACS, FCS2 and RICT was carried out for 11 species.

The analysis showed conclusively that one species, signal crayfish (*Pacifastacus leniusculus*), has an impact on measures of ecological quality. The invasion of sites by signal crayfish resulted in a lower EQR of NTAXA and a higher EQR of ASPT. It is possible that signal crayfish caused a lower EQR of macrophytes, although the evidence for this was less strong. However, invasive signal crayfish have been shown to cause significant reductions in the biomass and richness of aquatic macrophytes in ponds (Nyström et al. 2001).

The impact of signal crayfish on the EQR of ASPT may have been a consequence of selective predation on low scoring taxa (for example, molluscs, Oligochaetes, Chironomids). Selective predation on molluscs has been noted for signal and other (procambarid) invasive crayfish in ponds (Nyström et al. 2001, Dorn 2013) and suggested as an explanation for differences in community composition between invaded and uninvaded sites (Crawford et al. 2006, Mathers 2017, Turley et al. 2017). Nevertheless, the impact on the EQR of ASPT may have been an artefact created by signal crayfish being included in the scoring system used to derive ASPT. Whereas replacement of native white clawed crayfish (*Austropotamobius pallipes*) with invasive signal crayfish would not alter the EQR of ASPT, the addition of signal crayfish to a site that previously had no crayfish would result in an increased EQR of ASPT. Furthermore, loss of other taxa from invaded sites is likely to result in an arithmetic increase in average score as signal crayfish return a relatively high score.

Signal crayfish are not the only INNS vulnerable to such effects. Other INNS are included in scoring systems (either explicitly or under wider taxonomic groupings with related native species) and other tools are affected, such that this artefact is likely to obscure any biological effect of INNS on ecological quality. For instance, a replacement of native *Lemna minor* (RMNI = 8.8; Lake Macrophyte Nutrient Index (LMNI) = 8.52) by the invasive *Lemna minuta* (RMNI = 9.21; LMNI = 10) would lead to a decrease in EQR, whereas if the native species present before invasion was *Lemna gibba* (RMNI = 10; LMNI = 7.66), invasion by *L. minuta* would cause a decrease in the EQR of rivers but an increase in lakes. Yet in both these hypothetical cases, invasion by *L. minuta* leads to the loss of a single congeneric species. To reiterate the conclusions of Mathers et al. (2016b), care must be taken when interpreting biomonitoring indices when invasive species are present.

Nevertheless, NTAXA as a measure is more robust to the presence of INNS. The arithmetic consequence of the presence of an INNS without any biological impact is +1 (or 0 if belonging to a taxon already present). Furthermore, signal crayfish were associated with a substantially lower EQR of NTAXA, with the difference representing approximately half to three-quarters of a class. It is apparent that signal crayfish did have a substantial negative influence on this measure of ecological quality and this is likely to be caused by a real biological impact.

The evidence presented here indicates that the impact of signal crayfish becomes more pronounced with the length of time that the species has been present (Figure 4.14). This is likely to be true of all INNS and may have influenced the findings of this study. It is more likely that impacts would be detected for the species that invaded water bodies early in the period for which data were available. The duration of colonisation required before any impact can be detected will depend on how large the impact of the species is. However, this finding has operational implications as the WFD tools are not likely to detect any impact of INNS for some time after the initial invasion, by which time the INNS is likely to have established a substantial population and be harder to deal with.

There is also strong evidence that demon shrimp (*Dikerogammarus haemobaphes*) has an impact on the EQR of NTAXA, constraining the recovery that was seen across sites elsewhere, with the difference between invaded and uninvaded sites again representing approximately half to three-quarters of a class. Again, the impact appeared to become more pronounced with the length of time that the species was present (Figure 4.18). The presence of demon shrimp has been associated with a loss of native *Gammarus pulex* (Johns et al. 2018), which could explain the difference detected. However, the lack of recovery of sites invaded by demon shrimp detected here indicates a potentially more profound impact on macroinvertebrate communities, with demon shrimp either excluding other families that were driving recovery at sites where demon shrimp were absent, or causing an impact such that any gains were cancelled by other losses. Further analysis at the community level may be able to determine the nature of this impact.

The lack of recovery in sites invaded by demon shrimp suggests the presence of the species may counteract benefits of programmes of measures, at considerable cost to taxpayers and other stakeholders.

With the exception of common carp, the data for the remaining 8 INNS investigated show some evidence of a difference in measures of ecological quality where they were present. The strength of the evidence varies among the species, with more significant results coming from the reach year scale analyses, where attribution of cause and effect is less robust.

Common carp are included as one of the 23 species used by the FCS2 tool to derive EQR (WFD UKTAG 2008), which confounds interpretation of the presence of common carp on EQR. Similarly, many INNS are included in the list of invertebrate and macrophyte species used by the WFD tools to derive EQR (Table 4.21). Hence, interpreting any difference for these species is difficult.

Table 4.21 List of INNS considered in this project indicating those species (in bold) that are included (either explicitly or under wider taxonomic groupings) in the list of taxa used by the WFD tools to determine EQR

Macrophytes[1]	Invertebrates[2]	Fish[3]
Acorus calamus	***Astacus astacus***	*Ameiurus melas*
Aponogeton distachyos	***Astacus leptodactylus***	*Carassius auratus*
Azolla filiculoides	*Branchiura sowerbyi*	*Ctenopharyngodon idella*
Cabomba caroliniana	***Caecidotea communis***	***Cyprinus carpio***
Crassula helmsii	***Chelicorophium curvispinum***	*Lepomis gibbosus*
Egeria densa	*Corbicula fluminea*	*Leucaspius delineatus*
Eichhornia crassipes	*Cordylophora caspia*	*Leuciscus idus*
Elodea callitrichoides	***Crangonyx pseudogracilis***	*Oncorhynchus mykiss*
Elodea canadensis	*Dikerogammarus haemobaphes*	*Pseudorasbora parva*
Elodea nuttallii	*Dikerogammarus villosus*	*Rhodeus sericeus*
Hydrocotyle ranunculoides	*Dreissena bugensis*	*Salvelinus fontinalis*
Juncus ensifolius	***Dreissena polymorpha***	*Sander lucioperca*
Lagarosiphon major	*Eriocheir sinensis*	*Silurus glanis*
Lemna minuta	***Ferrissia (Petancyclus) wautieri/clessiniana***	
Ludwigia grandiflora	***Gammarus tigrinus***	
Ludwigia peploides	***Girardia tigrina / Dugesia tigrina***	
Lysichiton americanus	*Hemimysis anomala*	
Mimulus guttatus	*Hypania invalida*	
Mimulus guttatus x luteus	*Marstoniopsis insubrica*	
Mimulus luteus	***Menetus (Dilatata) dilatatus***	
Mimulus moschatus	***Musculium transversum***	
Mimulus ringens	***Mytilopsis leucophaeata***	
Myriophyllum aquaticum	*Orconectes limosus*	
Myriophyllum heterophyllum	*Orconectes virilis*	
Sagittaria latifolia	***Pacifastacus leniusculus***	
	Physella acuta	
	Physella gyrina	
	Planaria torva	
	Potamopyrgus antipodarum	
	Procambarus clarkii	
	Rangia cuneata	

Notes:
[1] WFD UKTAG (2014b, 2014c)
[2] WFD UKTAG (2014a)
[3] WFD UKTAG (2008)

4.6 Limitations

As the data used were not collected for the purpose of demonstrating an impact of INNS, as with all mensurative ‘experiments’ there are a number of limitations on the interpretation of the results. In all the analyses, the INNS were not manipulated to form the experimental treatments of ‘with’ and ‘without’. Instead sites were allocated to the 2 experimental categories based on the recorded presence or absence of the INNS. Hence, there is the possibility of a false positive (site allocated to the ‘with INNS’ group when it is actually absent from the site at the time when EQR was measured) or a false negative (site allocated to the ‘without INNS’ group when it is actually present at the site at the time when EQR was measured).

The probability of false positives and negatives is related to the spatial and temporal scales represented in the data. To deal with this, the analyses were made at 2 scales (reach year and water body WFD reporting period), with a higher confidence of not committing false positives or negatives at the reach year scale. Even at the reach year scale, however, there is the possibility that occurrence of an INNS at a low density could be missed during sampling as there is evidence to suggest that INNS can be present at a site for some time before they are first detected. This possibility cannot be discounted but, as impacts of INNS are likely to be density dependent, those sites where INNS were missed during sampling and falsely allocated to the ‘without’ group were unlikely to be suffering from substantial impacts. To account for such false negatives, the 2 groups should correctly be considered ‘with INNS at a density likely to be detected’ and ‘without INNS at a density likely to be detected’.

Allocation of sites to the ‘without INNS’ group is also important in terms of their representativeness of the control. This is fundamental for analysis at the water body WFD reporting period scale. If there was something unique to the group of water bodies allocated to the control such that they did not represent the generally expected trend with time, a false result would be returned where a difference was accepted as significant that was not real (Type II error). In the BACI analysis, care was taken to select as many control sites as possible to avoid such errors, although it was likely to have affected the results of the test for Himalayan balsam. Similarly, the ‘with INNS’ group should include multiple water bodies to avoid the influence of unique effects not associated with the presence of the INNS. It should be noted that any time series analysis without an adequate control group (see, for example, Turley et al. 2017) cannot confidently attribute cause to any change detected over time (Underwood 1992).

There are other factors affecting the probability of detecting a difference with a BACI analysis, including effect size and replication. As the impacts of INNS are likely to become more pronounced with time since first detection (press disturbance) as the population grows, effect size is influenced by when sites were first invaded. Those species that invaded a large number of water bodies early in the time series for which data were available were most likely to return significant results. The lack of a significant result for species that became widespread at a time before the data available, or late in the time series, or have only invaded a few water bodies, should not be interpreted as the absence of an impact for those species, merely that an impact could not be detected with the data available. This is also true for the EQR of macrophytes where no significant effects were detected using the BACI approach, largely due to limitations on the amount of data available.

Another influence on the BACI design is variability in the data, particularly in the control group. The probability of detecting a difference is dependent on temporal variation relative to effect size. The limited number of significant results for the EQR of fish was probably influenced by the highly variable EQR returned by FCS2, where individual water bodies spanned the full range of EQR during the time period for which data were available.

Finally, there are limitations on the interpretation of the results. The analyses undertaken at reach year scale were more likely to detect differences that are real, but can only show associations between INNS and differences, rather the demonstrate cause and effect. The BACI analysis undertaken at the water body WFD reporting period scale was less likely to be able to detect a significant difference that is real (that is, a higher probability of a Type I error). But where significant effects were detected, it does provide convincing evidence that the INNS has caused an impact on measures of ecological quality.

5 Conclusions and implications

The analysis performed in this project provides strong evidence that at least 2 of the INNS tested (signal crayfish and demon shrimp) have substantial impacts on the WFD measures of ecological quality. The impact of these species results in an effective reduction of EQR equivalent to approximately half to three-quarters of a WFD class. It is likely that other INNS do have an impact on the WFD measures of ecological quality, although the confidence in the evidence is less strong.

The evidence indicates that the impact of INNS increases with time after first detection. This is consistent with the biological understanding of how the impacts of INNS manifest themselves; impacts become more profound as the INNS population establishes and density increases. Although the time required before impacts are evident will depend on the scale of the impact, the evidence presented here suggests that significant differences are detectable more than 5 years after invasion. However, this finding has operational implications, as it is likely that the WFD tools will not detect any impact on INNS for some time after initial invasion, by which time the INNS is likely to have established a substantial population and be harder to deal with.

Understanding the mechanism by which INNS cause an impact on measures of ecological quality is confounded since many INNS are included in the list of taxa used to measure ecological quality (Table 4.21). The occurrence of an INNS may have a positive or negative arithmetic influence on the measure of ecological quality returned (depending on how they are perceived within the tool used to derive the measure), with neither effect based on a real biological impact on the quality of the site. Further effects on measures of ecological quality will arise where biological impacts of the INNS manifest themselves. However, their influence on measures of ecological quality will depend on the nature of the biological impact and how the INNS is perceived within the tool used to derive the measure. Such influences of INNS on measures of ecological quality will have operational implications, as the occurrence of INNS is likely to confound interpretation of other stressors, potentially leading to inappropriate programmes of measures.

The current system for assessing and classifying surface water bodies based on the presence of high impact alien species provides a procedure for downgrading sites where there is evidence that the INNS is causing more than a slight impact on a BQE (WFD UKTAG 2014d). This approach may lead to ‘double accounting’ for INNS where impacts are already apparent in the EQR returned, or a plus/minus effect where the tools are confounded (by including INNS in the taxa considered) such that the effect on EQR is positive (for example, for ASPT with signal crayfish).

6 Future work

The following investigations are suggested in order to obtain a fuller understanding of the impact of INNS on WFD measures of ecological quality.

6.1 Inclusion of more EQR data

The analyses to detect impacts would benefit from a longer period of EQR data. This could be achieved by including EQR data derived by the Environment Agency after 2014. It may also be possible to calculate EQR where community data and appropriate environmental data are available from dates before the EQR data used here. Longer time series of EQR data will provide the opportunity to test the impact of more species using the BACI approach (those INNS whose spread was prior to and late on in the time series used here) and provide a higher probability of detecting any impacts (including for those species include in the analyses here).

6.2 Impact of number of INNS

The study investigated the impact of individual species. However, the data found up to 25 INNS in a single water body and there is the possibility that the impact of the invasion of sites by multiple species will cause more profound and potentially non-additive impacts (‘invasion meltdown’). Further analyses to determine the impact of multiple invasions would provide important information for the Environment Agency on how best to manage such multiple invasion scenarios.

6.3 Influence of removing INNS from EQR assessments

All the WFD tools used to determine ecological quality investigated include INNS in the list of taxa used to derive the measure of EQR. It may be possible to exclude INNS from the measure of EQR in order to:

- determine the effect on the remaining community
- provide a cleaner signal of the impact of the species on the ecological quality of the site

Such an approach may provide an operational solution to the confounded effect of INNS on EQR.

6.4 Community level analysis

Data are available at the community level for the samples used to derive EQR, but were only used here to determine the presence or absence of INNS. These data could be analysed to determine what impacts INNS have at the community/species level. Such analysis could provide an alternative approach to assess the impact of INNS on ecological quality and thus form the basis of an alternative tool for assessing impacts of INNS. Any such analysis could be used to identify characteristic changes in communities associated with the presence of specific INNS, which could be used to attribute changes in measures of ecological quality to invasion by that INNS rather than other stressors.

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List of abbreviations

ANOVA	analysis of variance
ASPT	average score per taxon
BACI	before–after–control–impact [experimental design]
B/A	before and after [time periods]
BMWP	Biological Monitoring Working Party
BQE	biological quality element
EICAT	Environmental Impact Classification for Alien Taxa
EQR	Ecological Quality Ratio
FCS2	Fisheries Classification Scheme 2
GBNNSS	GB Non-Native Species Secretariat
GIS	geographical information system
INNS	invasive non-native species
LMNI	Lake Macrophyte Nutrient Index
MC	Minimal concern [EICAT grade]
MCP	minimum convex polygon
MN	Minor [EICAT grade]
MO	Moderate [EICAT grade]
MR	Major [EICAT grade]
MV	Massive [EICAT grade]
NTAXA	number of scoring taxa
OS	Ordnance Survey
Q_{90}	90th quantile
RICT	River Invertebrate Classification Tool
RMNI	River Macrophyte Nutrient Index
UC	Unclassified [EICAT grade]
WFD	Water Framework Directive
WFD UKTAG	Water Framework Directive – United Kingdom Technical Advisory Group
WHPT	Whalley, Hawkes, Paisley and Trigg metric

Appendix A: Area of extent of INNS

Table A.1 Area of extent for each INNS considered, calculated for all records and by decade

Name	Time period	Number			Area[1]			
		Records	GR	10km	10km	MCP	MCP INT	AHULL
Acorus calamus	All	1,139	889	329	32,900	150,912	118,791	102,026
Acorus calamus	1960s	0	0	0	0			
Acorus calamus	1970s	11	11	9	900	59,573	57,225	1,043
Acorus calamus	1980s	25	25	18	1,800	37,633	37,599	3,790
Acorus calamus	1990s	267	208	133	13,300	120,019	102,780	71,093
Acorus calamus	2000s	458	380	183	18,300	132,084	113,850	76,818
Acorus calamus	2010s	376	337	156	15,600	118,163	102,314	79,688
Ameiurus melas	All	1	1	1	100			
Ameiurus melas	1960s	0	0	0	0			
Ameiurus melas	1970s	0	0	0	0			
Ameiurus melas	1980s	1	1	1	100			
Ameiurus melas	1990s	0	0	0	0			
Ameiurus melas	2000s	0	0	0	0			
Ameiurus melas	2010s	0	0	0	0			
Aponogeton distachyos	All	85	64	46	4,600	131,313	95,322	17,074
Aponogeton distachyos	1960s	0	0	0	0			
Aponogeton distachyos	1970s	0	0	0	0			
Aponogeton distachyos	1980s	0	0	0	0			
Aponogeton distachyos	1990s	11	11	11	1,100	55,895	51,040	286
Aponogeton distachyos	2000s	44	37	27	2,700	131,102	95,163	3,481
Aponogeton distachyos	2010s	30	26	19	1,900	75,513	59,415	3,717
Astacus astacus	All	8	5	2	200	8	8	95
Astacus astacus	1960s	0	0	0	0			
Astacus astacus	1970s	0	0	0	0			
Astacus astacus	1980s	1	1	1	100			
Astacus astacus	1990s	3	2	1	100			
Astacus astacus	2000s	2	2	1	100			
Astacus astacus	2010s	0	0	0	0			
Astacus leptodactylus	All	176	71	40	4,000	73,074	66,255	10,885
Astacus leptodactylus	1960s	0	0	0	0			
Astacus leptodactylus	1970s	8	3	3	300	5,960	5,960	
Astacus leptodactylus	1980s	28	13	7	700	7,404	7,394	422
Astacus leptodactylus	1990s	106	38	24	2,400	34,717	34,645	5,888
Astacus leptodactylus	2000s	30	17	11	1,100	52,539	49,025	306
Astacus leptodactylus	2010s	2	2	1	100			
Azolla filiculoides	All	1920	1,510	471	47,100	190,672	124,434	108,889
Azolla filiculoides	1960s	2	2	1	100			
Azolla filiculoides	1970s	7	7	5	500	2,070	2,064	310
Azolla filiculoides	1980s	71	67	42	4,200	96,538	83,354	17,897
Azolla filiculoides	1990s	536	415	230	23,000	184,538	121,946	88,477
Azolla filiculoides	2000s	622	503	253	25,300	167,604	118,113	100,774
Azolla filiculoides	2010s	674	597	212	21,200	160,418	114,968	90,210
Branchiura sowerbyi	All	49	33	24	2,400	39,483	39,285	6,786
Branchiura sowerbyi	1960s	0	0	0	0			
Branchiura sowerbyi	1970s	0	0	0	0			
Branchiura sowerbyi	1980s	0	0	0	0			
Branchiura sowerbyi	1990s	1	1	1	100			
Branchiura sowerbyi	2000s	10	9	8	800	13,966	13,921	217
Branchiura sowerbyi	2010s	38	25	18	1,800	28,315	28,146	4,287

Name	Time period	Number			Area[1]			
		Records	GR	10km	10km	MCP	MCP INT	AHULL
Cabomba caroliniana	All	50	43	2	200	50	50	321
Cabomba caroliniana	1960s	0	0	0	0			
Cabomba caroliniana	1970s	0	0	0	0			
Cabomba caroliniana	1980s	0	0	0	0			
Cabomba caroliniana	1990s	0	0	0	0			
Cabomba caroliniana	2000s	18	14	2	200	9	9	170
Cabomba caroliniana	2010s	32	29	2	200	44	44	315
Caecidotea communis	All	4	4	4	400			1,462
Caecidotea communis	1960s	0	0	0	0			
Caecidotea communis	1970s	0	0	0	0			
Caecidotea communis	1980s	0	0	0	0			
Caecidotea communis	1990s	0	0	0	0			
Caecidotea communis	2000s	1	1	1	100			
Caecidotea communis	2010s	3	3	3	300			
Carassius auratus	All	257	208	139	13,900	126,044	105,998	67,717
Carassius auratus	1960s	0	0	0	0			
Carassius auratus	1970s	0	0	0	0			
Carassius auratus	1980s	28	27	21	2,100	29,405	27,281	4,852
Carassius auratus	1990s	80	56	45	4,500	108,553	95,433	11,576
Carassius auratus	2000s	102	93	76	7,600	72,576	70,058	47,569
Carassius auratus	2010s	44	42	37	3,700	98,132	81,831	15,378
Chelicorophium curvispinum	All	2,049	570	216	21,600	89,244	82,570	68,478
Chelicorophium curvispinum	1960s	0	0	0	0			
Chelicorophium curvispinum	1970s	18	2	1	100			
Chelicorophium curvispinum	1980s	134	65	45	4,500	30,469	29,104	15,388
Chelicorophium curvispinum	1990s	824	281	126	12,600	68,238	63,861	28,518
Chelicorophium curvispinum	2000s	394	130	67	6,700	62,018	58,492	32,345
Chelicorophium curvispinum	2010s	679	240	131	13,100	78,994	73,849	50,278
Claytonia sibirica	All	1,314	1,212	336	33,600	181,560	119,386	85,216
Claytonia sibirica	1960s	0	0	0	0			
Claytonia sibirica	1970s	66	66	27	2,700	70,365	50,363	7,331
Claytonia sibirica	1980s	43	42	19	1,900	28,236	12,409	2,941
Claytonia sibirica	1990s	301	282	117	11,700	133,300	100,643	45,680
Claytonia sibirica	2000s	475	437	180	18,000	179,878	118,701	57,625
Claytonia sibirica	2010s	429	402	184	18,400	158,590	112,286	73,687
Corbicula fluminea	All	189	51	31	3,100	47,033	46,109	6,231
Corbicula fluminea	1960s	0	0	0	0			
Corbicula fluminea	1970s	0	0	0	0			
Corbicula fluminea	1980s	0	0	0	0			
Corbicula fluminea	1990s	2	2	1	100			
Corbicula fluminea	2000s	68	20	10	1,000	19,207	18,864	748
Corbicula fluminea	2010s	119	36	26	2,600	46,528	45,632	3,256
Cordylophora caspia	All	20	19	12	1,200	66,827	61,914	1,378
Cordylophora caspia	1960s	0	0	0	0			
Cordylophora caspia	1970s	0	0	0	0			
Cordylophora caspia	1980s	11	10	4	400	49,146	46,664	223
Cordylophora caspia	1990s	7	7	6	600	1,075	442	666
Cordylophora caspia	2000s	1	1	1	100			
Cordylophora caspia	2010s	0	0	0	0			
Crangonyx pseudogracilis	All	28,369	7,551	1,102	110,200	187,691	129,333	128,088
Crangonyx pseudogracilis	1960s	0	0	0	0			
Crangonyx pseudogracilis	1970s	171	83	16	1,600	30,019	29,417	2,938
Crangonyx pseudogracilis	1980s	2,916	1,005	297	29,700	107,127	94,268	71,139
Crangonyx pseudogracilis	1990s	10,041	3,131	615	61,500	169,360	125,244	112,531
Crangonyx pseudogracilis	2000s	6,745	2,507	735	73,500	179,932	127,733	121,667
Crangonyx pseudogracilis	2010s	8,493	3,287	956	95,600	186,799	129,087	127,307
Crassula helmsii	All	2,894	2,387	645	64,500	183,633	126,912	121,703
Crassula helmsii	1960s	0	0	0	0			
Crassula helmsii	1970s	1	1	1	100			

Name	Time period	Number			Area[1]			
		Records	GR	10km	10km	MCP	MCP INT	AHULL
Crassula helmsii	1980s	9	8	8	800	44,120	42,344	201
Crassula helmsii	1990s	367	311	202	20,200	160,207	117,579	82,818
Crassula helmsii	2000s	1,238	1,066	396	39,600	178,690	126,018	114,246
Crassula helmsii	2010s	1,273	1,137	403	40,300	180,635	125,529	116,575
C. x crocosmiiflora	All	4,601	4,388	554	55,400	203,204	128,614	124,953
C. x crocosmiiflora	1960s	0	0	0	0			
C. x crocosmiiflora	1970s	8	8	8	800	20,961	12,314	406
C. x crocosmiiflora	1980s	18	18	7	700	3,901	3,458	813
C. x crocosmiiflora	1990s	579	536	168	16,800	166,634	106,720	61,242
C. x crocosmiiflora	2000s	2,298	2,214	346	34,600	201,891	128,551	113,130
C. x crocosmiiflora	2010s	1,684	1,657	342	34,200	190,318	122,381	95,046
Crocosmia paniculata	All	178	166	120	12,000	159,238	100,922	48,536
Crocosmia paniculata	1960s	0	0	0	0			
Crocosmia paniculata	1970s	0	0	0	0			
Crocosmia paniculata	1980s	0	0	0	0			
Crocosmia paniculata	1990s	25	24	21	2,100	88,222	61,404	4,027
Crocosmia paniculata	2000s	67	61	50	5,000	136,452	81,088	20,680
Crocosmia paniculata	2010s	86	83	62	6,200	139,803	96,432	26,528
Ctenopharyngodon idella	All	45	40	34	3,400	65,326	62,730	17,362
Ctenopharyngodon idella	1960s	0	0	0	0			
Ctenopharyngodon idella	1970s	0	0	0	0			
Ctenopharyngodon idella	1980s	6	6	6	600	6,166	6,165	693
Ctenopharyngodon idella	1990s	12	10	9	900	39,229	38,972	1,205
Ctenopharyngodon idella	2000s	24	23	20	2,000	42,104	40,796	6,828
Ctenopharyngodon idella	2010s	3	3	3	300			
Cyprinus carpio	All	2,703	1,868	606	60,600	155,621	119,572	109,249
Cyprinus carpio	1960s	0	0	0	0			
Cyprinus carpio	1970s	7	7	7	700	18,581	18,537	298
Cyprinus carpio	1980s	211	168	112	11,200	73,166	69,809	43,063
Cyprinus carpio	1990s	909	672	305	30,500	116,035	104,596	80,519
Cyprinus carpio	2000s	1,274	973	420	42,000	152,083	118,714	101,420
Cyprinus carpio	2010s	301	208	139	13,900	90,437	84,382	61,702
Dikerogammarus haemobaphes	All	782	364	178	17,800	61,963	61,681	50,341
Dikerogammarus haemobaphes	1960s	0	0	0	0			
Dikerogammarus haemobaphes	1970s	0	0	0	0			
Dikerogammarus haemobaphes	1980s	0	0	0	0			
Dikerogammarus haemobaphes	1990s	0	0	0	0			
Dikerogammarus haemobaphes	2000s	0	0	0	0			
Dikerogammarus haemobaphes	2010s	782	364	178	17,800	61,963	61,681	50,341
Dikerogammarus villosus	All	50	18	8	800	24,931	20,652	446
Dikerogammarus villosus	1960s	0	0	0	0			
Dikerogammarus villosus	1970s	0	0	0	0			
Dikerogammarus villosus	1980s	0	0	0	0			
Dikerogammarus villosus	1990s	0	0	0	0			
Dikerogammarus villosus	2000s	0	0	0	0			
Dikerogammarus villosus	2010s	50	18	8	800	24,931	20,652	446
Dreissena bugensis	All	19	16	6	600	443	434	845
Dreissena bugensis	1960s	0	0	0	0			
Dreissena bugensis	1970s	0	0	0	0			
Dreissena bugensis	1980s	0	0	0	0			
Dreissena bugensis	1990s	0	0	0	0			
Dreissena bugensis	2000s	0	0	0	0			
Dreissena bugensis	2010s	19	16	6	600	443	434	845
Dreissena polymorpha	All	1,795	610	248	24,800	122,069	102,543	67,709
Dreissena polymorpha	1960s	19	18	17	1,700	48,855	45,090	12,571
Dreissena polymorpha	1970s	36	33	26	2,600	36,449	36,356	7,327
Dreissena polymorpha	1980s	103	57	35	3,500	40,236	39,685	21,326
Dreissena polymorpha	1990s	512	128	71	7,100	63,002	61,585	36,554
Dreissena polymorpha	2000s	673	253	147	14,700	109,862	93,238	57,191

Name	Time period	Number			Area[1]			
		Records	GR	10km	10km	MCP	MCP INT	AHULL
Dreissena polymorpha	2010s	440	251	136	13,600	102,753	92,782	53,010
Egeria densa	All	65	55	46	4,600	155,844	110,745	10,777
Egeria densa	1960s	0	0	0	0			
Egeria densa	1970s	1	1	1	100			
Egeria densa	1980s	0	0	0	0			
Egeria densa	1990s	8	7	7	700	9,734	9,292	1,055
Egeria densa	2000s	31	28	22	2,200	143,347	101,717	3,105
Egeria densa	2010s	25	21	21	2,100	112,759	89,758	4,437
Eichhornia crassipes	All	12	12	10	1,000	54,204	48,230	1,063
Eichhornia crassipes	1960s	0	0	0	0			
Eichhornia crassipes	1970s	0	0	0	0			
Eichhornia crassipes	1980s	0	0	0	0			
Eichhornia crassipes	1990s	2	2	1	100			
Eichhornia crassipes	2000s	7	7	6	600	28,866	23,705	481
Eichhornia crassipes	2010s	3	3	3	300			
Elodea callitrichoides	All	3	2	2	200			
Elodea callitrichoides	1960s	0	0	0	0			
Elodea callitrichoides	1970s	0	0	0	0			
Elodea callitrichoides	1980s	0	0	0	0			
Elodea callitrichoides	1990s	1	1	1	100			
Elodea callitrichoides	2000s	1	1	1	100			
Elodea callitrichoides	2010s	0	0	0	0			
Elodea canadensis	All	5,373	3,712	880	88,000	187,922	129,271	128,059
Elodea canadensis	1960s	6	6	5	500	1,762	1,457	524
Elodea canadensis	1970s	241	239	155	15,500	142,062	110,454	54,491
Elodea canadensis	1980s	669	553	256	25,600	154,172	121,364	104,086
Elodea canadensis	1990s	1,671	1,249	462	46,200	177,243	127,558	117,831
Elodea canadensis	2000s	1,605	1,192	494	49,400	182,157	128,090	123,435
Elodea canadensis	2010s	1,150	875	424	42,400	173,442	127,458	119,626
Elodea nuttallii	All	8,546	4,923	785	78,500	187,598	129,224	124,895
Elodea nuttallii	1960s	0	0	0	0			
Elodea nuttallii	1970s	42	42	29	2,900	29,223	28,258	6,147
Elodea nuttallii	1980s	244	217	107	10,700	97,356	91,229	39,414
Elodea nuttallii	1990s	2,332	1,323	404	40,400	148,621	118,542	112,162
Elodea nuttallii	2000s	2,736	1,837	510	51,000	183,978	128,472	119,078
Elodea nuttallii	2010s	3,148	2,270	567	56,700	183,557	128,201	118,367
Eriocheir sinensis	All	302	191	68	6,800	102,457	92,071	26,396
Eriocheir sinensis	1960s	0	0	0	0			
Eriocheir sinensis	1970s	0	0	0	0			
Eriocheir sinensis	1980s	2	2	2	200			
Eriocheir sinensis	1990s	2	2	2	200			
Eriocheir sinensis	2000s	68	28	21	2,100	45,172	43,374	5,423
Eriocheir sinensis	2010s	229	162	57	5,700	100,723	90,995	24,345
Fallopia japonica	All	14,781	12,455	1,069	106,900	204,091	130,037	129,608
Fallopia japonica	1960s	3	3	2	200			
Fallopia japonica	1970s	23	23	15	1,500	6,080	5,880	3,226
Fallopia japonica	1980s	237	222	44	4,400	55,972	53,096	6,175
Fallopia japonica	1990s	3,269	2,702	547	54,700	198,742	128,383	121,350
Fallopia japonica	2000s	4,029	3,454	667	66,700	203,559	129,899	126,125
Fallopia japonica	2010s	7,166	6,513	866	86,600	203,402	129,918	128,206
Fallopia sachalinensis	All	240	220	150	15,000	147,750	115,347	69,749
Fallopia sachalinensis	1960s	0	0	0	0			
Fallopia sachalinensis	1970s	0	0	0	0			
Fallopia sachalinensis	1980s	0	0	0	0			
Fallopia sachalinensis	1990s	69	66	53	5,300	101,643	91,098	27,877
Fallopia sachalinensis	2000s	93	86	69	6,900	127,748	103,745	39,346
Fallopia sachalinensis	2010s	78	72	58	5,800	92,121	83,329	32,547
Fallopia x bohemica	All	223	204	115	11,500	153,845	117,742	50,341

Name	Time period	Number			Area[1]			
		Records	**GR**	**10km**	**10km**	**MCP**	**MCP INT**	**AHULL**
Fallopia x bohemica	1960s	0	0	0	0			
Fallopia x bohemica	1970s	0	0	0	0			
Fallopia x bohemica	1980s	0	0	0	0			
Fallopia x bohemica	1990s	95	84	56	5,600	137,563	108,654	22,877
Fallopia x bohemica	2000s	75	70	48	4,800	119,912	91,756	14,916
Fallopia x bohemica	2010s	53	52	36	3,600	65,674	64,891	13,450
Ferrissia (Petancyclus) wautieri	All	319	226	139	13,900	116,410	93,084	59,382
Ferrissia (Petancyclus) wautieri	1960s	0	0	0	0			
Ferrissia (Petancyclus) wautieri	1970s	9	4	4	400	10,432	10,265	
Ferrissia (Petancyclus) wautieri	1980s	23	10	8	800	27,783	27,030	3,411
Ferrissia (Petancyclus) wautieri	1990s	24	13	10	1,000	19,683	17,536	1,886
Ferrissia (Petancyclus) wautieri	2000s	126	106	77	7,700	105,051	86,317	48,629
Ferrissia (Petancyclus) wautieri	2010s	135	101	69	6,900	74,052	72,065	42,614
Gammarus tigrinus	All	3,230	691	219	21,900	106,554	88,056	48,406
Gammarus tigrinus	1960s	0	0	0	0			
Gammarus tigrinus	1970s	7	1	1	100			
Gammarus tigrinus	1980s	791	191	71	7,100	56,484	51,050	21,709
Gammarus tigrinus	1990s	1,990	473	143	14,300	47,171	46,164	31,967
Gammarus tigrinus	2000s	150	82	51	5,100	30,787	30,387	19,210
Gammarus tigrinus	2010s	292	155	99	9,900	87,922	73,843	40,253
Girardia tigrina	All	3,084	1,275	545	54,500	159,488	122,169	115,691
Girardia tigrina	1960s	0	0	0	0			
Girardia tigrina	1970s	0	0	0	0			
Girardia tigrina	1980s	7	7	7	700	24,303	18,738	749
Girardia tigrina	1990s	349	239	141	14,100	123,107	107,883	48,675
Girardia tigrina	2000s	1,217	616	310	31,000	147,193	113,511	100,514
Girardia tigrina	2010s	1,510	707	390	39,000	151,712	119,180	110,343
Hemimysis anomala	All	52	47	42	4,200	29,965	29,965	24,803
Hemimysis anomala	1960s	0	0	0	0			
Hemimysis anomala	1970s	0	0	0	0			
Hemimysis anomala	1980s	0	0	0	0			
Hemimysis anomala	1990s	0	0	0	0			
Hemimysis anomala	2000s	4	4	4	400			935
Hemimysis anomala	2010s	48	44	40	4,000	29,965	29,965	21,563
Heracleum mantegazzianum	All	3,997	3,483	733	73,300	178,075	128,201	124,946
Heracleum mantegazzianum	1960s	2	2	1	100			
Heracleum mantegazzianum	1970s	12	12	11	1,100	56,480	47,217	2,316
Heracleum mantegazzianum	1980s	54	51	25	2,500	69,399	68,041	3,682
Heracleum mantegazzianum	1990s	771	571	332	33,200	156,185	121,472	109,938
Heracleum mantegazzianum	2000s	967	883	396	39,600	158,503	123,236	116,557
Heracleum mantegazzianum	2010s	2,186	2,039	433	43,300	174,759	127,258	115,119
Hydrocotyle ranunculoides	All	976	826	180	18,000	154,756	110,156	78,552
Hydrocotyle ranunculoides	1960s	0	0	0	0			
Hydrocotyle ranunculoides	1970s	0	0	0	0			
Hydrocotyle ranunculoides	1980s	0	0	0	0			
Hydrocotyle ranunculoides	1990s	51	43	24	2,400	44,011	42,348	9,246
Hydrocotyle ranunculoides	2000s	263	220	83	8,300	150,618	107,490	42,882
Hydrocotyle ranunculoides	2010s	662	591	131	13,100	109,706	97,287	59,502
Hypania invalida	All	137	46	28	2,800	31,910	31,901	6,990
Hypania invalida	1960s	0	0	0	0			
Hypania invalida	1970s	0	0	0	0			
Hypania invalida	1980s	0	0	0	0			
Hypania invalida	1990s	0	0	0	0			
Hypania invalida	2000s	4	2	1	100			
Hypania invalida	2010s	133	46	28	2,800	31,910	31,901	6,990
Impatiens capensis	All	2,207	1,848	310	31,000	118,145	99,867	72,452
Impatiens capensis	1960s	0	0	0	0			
Impatiens capensis	1970s	17	17	12	1,200	13,426	13,423	1,069
Impatiens capensis	1980s	41	40	23	2,300	22,969	22,969	9,507

Name	Time period	Number			Area^1			
		Records	GR	10km	10km	MCP	MCP INT	AHULL
Impatiens capensis	1990s	265	254	116	11,600	66,361	59,813	39,867
Impatiens capensis	2000s	654	485	179	17,900	115,497	98,312	54,238
Impatiens capensis	2010s	1,228	1,125	244	24,400	105,607	91,946	59,131
Impatiens glandulifera	All	27,311	22,132	1,140	114,000	192,610	129,845	129,297
Impatiens glandulifera	1960s	4	4	4	400			
Impatiens glandulifera	1970s	100	100	66	6,600	116,300	89,171	24,750
Impatiens glandulifera	1980s	348	338	115	11,500	138,044	112,861	43,940
Impatiens glandulifera	1990s	4,574	3,413	687	68,700	185,655	128,422	117,128
Impatiens glandulifera	2000s	6,126	5,295	876	87,600	188,993	129,275	127,321
Impatiens glandulifera	2010s	16,011	13,861	1,003	100,300	192,057	129,811	128,008
Impatiens parviflora	All	836	741	216	21,600	145,595	117,309	76,691
Impatiens parviflora	1960s	0	0	0	0			
Impatiens parviflora	1970s	5	5	5	500	2,284	2,284	374
Impatiens parviflora	1980s	17	17	6	600	36,717	35,919	315
Impatiens parviflora	1990s	117	107	65	6,500	87,706	84,090	25,917
Impatiens parviflora	2000s	271	252	129	12,900	137,842	113,020	59,969
Impatiens parviflora	2010s	425	400	108	10,800	102,746	96,931	54,132
Juncus ensifolius	All	8	7	4	400	7,982	7,982	212
Juncus ensifolius	1960s	0	0	0	0			
Juncus ensifolius	1970s	0	0	0	0			
Juncus ensifolius	1980s	0	0	0	0			
Juncus ensifolius	1990s	0	0	0	0			
Juncus ensifolius	2000s	5	4	2	200	24	24	363
Juncus ensifolius	2010s	3	3	2	200			
Lagarosiphon major	All	739	647	338	33,800	168,903	114,506	109,110
Lagarosiphon major	1960s	0	0	0	0			
Lagarosiphon major	1970s	0	0	0	0			
Lagarosiphon major	1980s	4	4	4	400			
Lagarosiphon major	1990s	261	219	135	13,500	122,450	96,780	72,791
Lagarosiphon major	2000s	263	246	162	16,200	164,132	112,114	87,073
Lagarosiphon major	2010s	207	192	133	13,300	147,783	111,304	72,337
Lemna minuta	All	4,219	3,297	780	78,000	184,586	124,075	121,139
Lemna minuta	1960s	0	0	0	0			
Lemna minuta	1970s	0	0	0	0			
Lemna minuta	1980s	6	6	5	500	14,327	14,327	
Lemna minuta	1990s	395	358	189	18,900	173,882	120,415	78,747
Lemna minuta	2000s	2,139	1,689	557	55,700	175,229	119,984	107,629
Lemna minuta	2010s	1,674	1,450	527	52,700	182,501	123,232	106,447
Lepomis gibbosus	All	31	30	10	1,000	84,138	51,149	907
Lepomis gibbosus	1960s	0	0	0	0			
Lepomis gibbosus	1970s	0	0	0	0			
Lepomis gibbosus	1980s	0	0	0	0			
Lepomis gibbosus	1990s	12	12	6	600	1,800	1,800	560
Lepomis gibbosus	2000s	17	16	6	600	84,134	51,146	443
Lepomis gibbosus	2010s	2	2	2	200			
Leucaspius delineatus	All	105	67	18	1,800	41,302	40,014	1,820
Leucaspius delineatus	1960s	0	0	0	0			
Leucaspius delineatus	1970s	0	0	0	0			
Leucaspius delineatus	1980s	0	0	0	0			
Leucaspius delineatus	1990s	34	33	11	1,100	1,716	1,716	867
Leucaspius delineatus	2000s	36	22	12	1,200	17,159	17,159	907
Leucaspius delineatus	2010s	35	20	10	1,000	7,697	7,681	777
Leuciscus idus	All	86	72	52	5,200	87,847	83,189	28,027
Leuciscus idus	1960s	0	0	0	0			
Leuciscus idus	1970s	0	0	0	0			
Leuciscus idus	1980s	6	5	5	500	4,263	3,829	610
Leuciscus idus	1990s	14	13	11	1,100	67,583	65,995	2,325
Leuciscus idus	2000s	49	44	34	3,400	67,973	65,490	14,743

Name	Time period	Number			Area[1]			
		Records	GR	10km	10km	MCP	MCP INT	AHULL
Leuciscus idus	2010s	16	12	9	900	22,280	22,187	703
Ludwigia grandiflora	All	54	37	24	2,400	106,975	84,673	12,543
Ludwigia grandiflora	1960s	0	0	0	0			
Ludwigia grandiflora	1970s	0	0	0	0			
Ludwigia grandiflora	1980s	0	0	0	0			
Ludwigia grandiflora	1990s	1	1	1	100			
Ludwigia grandiflora	2000s	26	20	11	1,100	58,133	56,753	1,177
Ludwigia grandiflora	2010s	27	23	18	1,800	35,431	30,402	11,302
Ludwigia peploides	All	8	7	3	300	14	14	63
Ludwigia peploides	1960s	0	0	0	0			
Ludwigia peploides	1970s	0	0	0	0			
Ludwigia peploides	1980s	0	0	0	0			
Ludwigia peploides	1990s	0	0	0	0			
Ludwigia peploides	2000s	8	7	3	300	14	14	63
Ludwigia peploides	2010s	0	0	0	0			
Lupinus nootkatensis	All	1	1	1	100			
Lupinus nootkatensis	1960s	0	0	0	0			
Lupinus nootkatensis	1970s	0	0	0	0			
Lupinus nootkatensis	1980s	0	0	0	0			
Lupinus nootkatensis	1990s	1	1	1	100			
Lupinus nootkatensis	2000s	0	0	0	0			
Lupinus nootkatensis	2010s	0	0	0	0			
Lysichiton americanus	All	849	745	213	21,300	172,604	123,498	84,891
Lysichiton americanus	1960s	0	0	0	0			
Lysichiton americanus	1970s	0	0	0	0			
Lysichiton americanus	1980s	0	0	0	0			
Lysichiton americanus	1990s	61	54	42	4,200	87,542	69,545	12,135
Lysichiton americanus	2000s	235	215	92	9,200	154,842	110,074	32,006
Lysichiton americanus	2010s	553	506	170	17,000	167,345	121,186	73,673
Marstoniopsis insubrica	All	18	7	6	600	11,409	11,409	111
Marstoniopsis insubrica	1960s	2	1	1	100			
Marstoniopsis insubrica	1970s	6	2	2	200			
Marstoniopsis insubrica	1980s	8	2	1	100			
Marstoniopsis insubrica	1990s	0	0	0	0			
Marstoniopsis insubrica	2000s	2	2	2	200			
Marstoniopsis insubrica	2010s	0	0	0	0			
Menetus (Dilatata) dilatatus	All	66	52	39	3,900	86,103	78,021	27,794
Menetus (Dilatata) dilatatus	1960s	2	1	1	100			
Menetus (Dilatata) dilatatus	1970s	20	12	5	500	1,703	1,703	308
Menetus (Dilatata) dilatatus	1980s	6	2	2	200			
Menetus (Dilatata) dilatatus	1990s	10	9	8	800	37,313	29,974	278
Menetus (Dilatata) dilatatus	2000s	12	12	11	1,100	42,871	42,794	2,538
Menetus (Dilatata) dilatatus	2010s	16	16	16	1,600	24,905	24,899	11,846
Mimulus guttatus/luteus grp	All	1,955	1,549	559	55,900	185,642	128,249	121,325
Mimulus guttatus/luteus grp	1960s	1	1	1	100			
Mimulus guttatus/luteus grp	1970s	121	118	76	7,600	133,897	106,801	16,903
Mimulus guttatus/luteus grp	1980s	99	95	65	6,500	123,573	104,827	17,942
Mimulus guttatus/luteus grp	1990s	370	326	188	18,800	146,844	116,861	71,494
Mimulus guttatus/luteus grp	2000s	639	454	259	25,900	181,260	126,918	106,014
Mimulus guttatus/luteus grp	2010s	720	637	343	34,300	177,135	126,412	96,132
Mimulus moschatus	All	134	120	79	7,900	144,279	108,312	39,209
Mimulus moschatus	1960s	1	1	1	100			
Mimulus moschatus	1970s	5	5	4	400	1,753	1,708	2,941
Mimulus moschatus	1980s	8	8	5	500	1,814	1,567	502
Mimulus moschatus	1990s	30	29	27	2,700	85,777	76,064	6,773
Mimulus moschatus	2000s	55	48	34	3,400	136,561	101,291	12,849
Mimulus moschatus	2010s	34	32	21	2,100	69,008	66,018	4,387
Musculium transversum	All	45	34	28	2,800	61,766	61,145	17,680

Name	Time period	Number			Area[1]			
		Records	GR	10km	10km	MCP	MCP INT	AHULL
Musculium transversum	1960s	3	3	3	300			
Musculium transversum	1970s	15	8	6	600	7,194	7,194	515
Musculium transversum	1980s	1	1	1	100			
Musculium transversum	1990s	5	4	2	200	68	65	74
Musculium transversum	2000s	9	9	9	900	55,701	55,426	914
Musculium transversum	2010s	12	10	10	1,000	26,279	26,144	633
Myriophyllum aquaticum	All	807	694	306	30,600	174,497	118,524	98,742
Myriophyllum aquaticum	1960s	1	1	1	100			
Myriophyllum aquaticum	1970s	0	0	0	0			
Myriophyllum aquaticum	1980s	0	0	0	0			
Myriophyllum aquaticum	1990s	193	159	113	11,300	128,209	95,949	66,310
Myriophyllum aquaticum	2000s	381	346	179	17,900	167,452	115,360	75,579
Myriophyllum aquaticum	2010s	230	215	137	13,700	144,395	109,749	68,194
Mytilopsis leucophaeata	All	27	9	5	500	2,791	2,789	
Mytilopsis leucophaeata	1960s	0	0	0	0			
Mytilopsis leucophaeata	1970s	0	0	0	0			
Mytilopsis leucophaeata	1980s	0	0	0	0			
Mytilopsis leucophaeata	1990s	0	0	0	0			
Mytilopsis leucophaeata	2000s	11	2	2	200			
Mytilopsis leucophaeata	2010s	16	9	5	500	2,791	2,789	623
Oncorhynchus mykiss	All	1,135	833	335	33,500	169,791	123,068	115,482
Oncorhynchus mykiss	1960s	1	1	1	100			
Oncorhynchus mykiss	1970s	6	5	5	500	14,241	14,220	
Oncorhynchus mykiss	1980s	43	42	28	2,800	62,129	58,460	11,344
Oncorhynchus mykiss	1990s	440	361	190	19,000	162,437	118,394	101,065
Oncorhynchus mykiss	2000s	476	363	202	20,200	148,293	117,639	89,522
Oncorhynchus mykiss	2010s	169	117	71	7,100	104,256	90,916	44,831
Orconectes limosus	All	8	7	6	600	27,375	27,125	174
Orconectes limosus	1960s	0	0	0	0			
Orconectes limosus	1970s	0	0	0	0			
Orconectes limosus	1980s	0	0	0	0			
Orconectes limosus	1990s	0	0	0	0			
Orconectes limosus	2000s	8	7	6	600	27,375	27,125	174
Orconectes limosus	2010s	0	0	0	0			
Orconectes virilis	All	3	3	2	200			
Orconectes virilis	1960s	0	0	0	0			
Orconectes virilis	1970s	0	0	0	0			
Orconectes virilis	1980s	0	0	0	0			
Orconectes virilis	1990s	0	0	0	0			
Orconectes virilis	2000s	3	3	2	200			
Orconectes virilis	2010s	0	0	0	0			
Pacifastacus leniusculus	All	3,682	1,753	477	47,700	172,031	124,015	109,604
Pacifastacus leniusculus	1960s	0	0	0	0			
Pacifastacus leniusculus	1970s	12	12	11	1,100	12,543	12,539	4,473
Pacifastacus leniusculus	1980s	145	116	72	7,200	95,196	87,420	46,072
Pacifastacus leniusculus	1990s	706	433	176	17,600	119,985	102,546	73,535
Pacifastacus leniusculus	2000s	1,286	649	241	24,100	111,517	101,411	83,578
Pacifastacus leniusculus	2010s	1,403	722	297	29,700	148,453	117,385	90,169
Petasites albus	All	70	58	38	3,800	91,412	86,529	22,369
Petasites albus	1960s	0	0	0	0			
Petasites albus	1970s	3	2	1	100			
Petasites albus	1980s	0	0	0	0			
Petasites albus	1990s	16	15	13	1,300	34,716	34,617	4,077
Petasites albus	2000s	27	22	19	1,900	79,234	76,540	7,085
Petasites albus	2010s	23	21	18	1,800	64,640	62,019	12,267
Petasites fragrans	All	3,305	3,065	597	59,700	202,911	129,105	124,487
Petasites fragrans	1960s	0	0	0	0			
Petasites fragrans	1970s	19	16	13	1,300	5,141	4,905	1,985

Name	Time period	Number			Area[1]			
		Records	GR	10km	10km	MCP	MCP INT	AHULL
Petasites fragrans	1980s	31	31	15	1,500	9,268	8,318	2,730
Petasites fragrans	1990s	377	354	153	15,300	172,328	117,239	64,071
Petasites fragrans	2000s	1,588	1,494	331	33,100	190,064	124,094	97,021
Petasites fragrans	2010s	1,284	1,235	376	37,600	200,720	128,055	119,389
Petasites japonicus	All	173	146	86	8,600	149,979	115,294	59,327
Petasites japonicus	1960s	0	0	0	0			
Petasites japonicus	1970s	2	2	2	200			
Petasites japonicus	1980s	1	1	1	100			
Petasites japonicus	1990s	44	40	26	2,600	128,384	102,899	6,803
Petasites japonicus	2000s	62	56	44	4,400	114,562	92,165	20,430
Petasites japonicus	2010s	64	59	45	4,500	137,229	107,673	19,657
Physella	All	2,914	1,644	561	56,100	193,822	127,534	113,325
Physella	1960s	1	1	1	100			
Physella	1970s	13	13	11	1,100	14,432	13,935	1,054
Physella	1980s	33	33	24	2,400	60,068	53,765	4,796
Physella	1990s	163	142	79	7,900	119,356	99,453	33,801
Physella	2000s	965	642	262	26,200	169,227	118,655	89,138
Physella	2010s	1,736	971	431	43,100	173,865	125,919	106,327
Planaria torva	All	671	469	313	31,300	161,476	124,138	110,396
Planaria torva	1960s	1	1	1	100			
Planaria torva	1970s	1	1	1	100			
Planaria torva	1980s	2	2	2	200			
Planaria torva	1990s	76	68	50	5,000	126,603	99,630	22,854
Planaria torva	2000s	352	231	164	16,400	150,305	115,988	82,469
Planaria torva	2010s	235	204	167	16,700	152,033	120,209	94,870
Potamopyrgus antipodarum	All	99,012	19,309	1,349	134,900	200,399	130,051	130,121
Potamopyrgus antipodarum	1960s	57	45	33	3,300	75,934	73,716	8,218
Potamopyrgus antipodarum	1970s	1,303	445	189	18,900	110,961	97,250	68,728
Potamopyrgus antipodarum	1980s	10,027	3,351	668	66,800	166,041	125,360	112,982
Potamopyrgus antipodarum	1990s	48,694	11,268	973	97,300	185,307	128,058	125,035
Potamopyrgus antipodarum	2000s	18,420	5,647	1,113	111,300	198,282	129,721	129,049
Potamopyrgus antipodarum	2010s	20,483	6,068	1,207	120,700	190,458	129,733	129,667
Procambarus clarkii	All	11	9	1	100	16	16	197
Procambarus clarkii	1960s	0	0	0	0			
Procambarus clarkii	1970s	0	0	0	0			
Procambarus clarkii	1980s	0	0	0	0			
Procambarus clarkii	1990s	2	1	1	100			
Procambarus clarkii	2000s	8	7	1	100	3	3	124
Procambarus clarkii	2010s	1	1	1	100			
Pseudorasbora parva	All	17	15	6	600	24,144	24,144	152
Pseudorasbora parva	1960s	0	0	0	0			
Pseudorasbora parva	1970s	0	0	0	0			
Pseudorasbora parva	1980s	0	0	0	0			
Pseudorasbora parva	1990s	0	0	0	0			
Pseudorasbora parva	2000s	8	8	5	500	24,144	24,144	130
Pseudorasbora parva	2010s	9	8	2	200	1	1	106
Rangia cuneata	All	5	5	2	200	4	4	177
Rangia cuneata	1960s	0	0	0	0			
Rangia cuneata	1970s	0	0	0	0			
Rangia cuneata	1980s	0	0	0	0			
Rangia cuneata	1990s	0	0	0	0			
Rangia cuneata	2000s	0	0	0	0			
Rangia cuneata	2010s	5	5	2	200	4	4	177
Rhodeus sericeus	All	299	123	24	2,400	11,654	11,634	2,915
Rhodeus sericeus	1960s	0	0	0	0			
Rhodeus sericeus	1970s	2	2	2	200			
Rhodeus sericeus	1980s	15	13	2	200	46	46	265
Rhodeus sericeus	1990s	93	51	9	900	3,390	3,390	746
Rhodeus sericeus	2000s	131	53	20	2,000	11,027	11,011	2,546

Name	**Time period**	**Number**			**Area[1]**			
		Records	**GR**	**10km**	**10km**	**MCP**	**MCP INT**	**AHULL**
Rhodeus sericeus	2010s	58	44	13	1,300	1,001	1,000	1,610
Rhododendron luteum	All	153	142	71	7,100	122,592	103,715	24,150
Rhododendron luteum	1960s	0	0	0	0			
Rhododendron luteum	1970s	0	0	0	0			
Rhododendron luteum	1980s	0	0	0	0			
Rhododendron luteum	1990s	30	27	21	2,100	92,640	86,159	2,531
Rhododendron luteum	2000s	44	42	35	3,500	94,194	82,747	13,820
Rhododendron luteum	2010s	79	76	37	3,700	69,362	68,935	8,904
Rhododendron ponticum	All	6,460	5,661	722	72,200	198,647	128,314	124,573
Rhododendron ponticum	1960s	16	15	10	1,000	5,630	5,146	2,270
Rhododendron ponticum	1970s	102	102	22	2,200	46,051	44,438	3,582
Rhododendron ponticum	1980s	448	410	43	4,300	44,973	43,009	7,170
Rhododendron ponticum	1990s	1,369	1,276	269	26,900	182,268	119,880	74,253
Rhododendron ponticum	2000s	2,015	1,821	436	43,600	194,724	127,445	114,458
Rhododendron ponticum	2010s	2,502	2,376	470	47,000	195,043	126,964	111,763
Sagittaria latifolia	All	73	59	41	4,100	123,828	87,296	7,716
Sagittaria latifolia	1960s	0	0	0	0			
Sagittaria latifolia	1970s	0	0	0	0			
Sagittaria latifolia	1980s	0	0	0	0			
Sagittaria latifolia	1990s	15	15	11	1,100	12,044	10,507	1,958
Sagittaria latifolia	2000s	26	20	17	1,700	101,549	66,247	2,749
Sagittaria latifolia	2010s	31	28	22	2,200	57,240	56,764	3,037
Salvelinus fontinalis	All	5	5	5	500	20,508	18,212	335
Salvelinus fontinalis	1960s	0	0	0	0			
Salvelinus fontinalis	1970s	0	0	0	0			
Salvelinus fontinalis	1980s	1	1	1	100			
Salvelinus fontinalis	1990s	3	3	3	300			
Salvelinus fontinalis	2000s	1	1	1	100			
Salvelinus fontinalis	2010s	0	0	0	0			
Sander lucioperca	All	825	404	77	7,700	32,865	32,486	28,258
Sander lucioperca	1960s	0	0	0	0			
Sander lucioperca	1970s	0	0	0	0			
Sander lucioperca	1980s	97	92	27	2,700	11,444	11,442	3,037
Sander lucioperca	1990s	463	264	53	5,300	24,322	24,295	11,204
Sander lucioperca	2000s	205	110	43	4,300	24,324	23,970	14,082
Sander lucioperca	2010s	60	44	27	2,700	24,581	24,546	13,965
Silurus glanis	All	11	9	9	900	43,327	41,751	1,307
Silurus glanis	1960s	0	0	0	0			
Silurus glanis	1970s	0	0	0	0			
Silurus glanis	1980s	0	0	0	0			
Silurus glanis	1990s	4	4	4	400			431
Silurus glanis	2000s	7	5	5	500	28,166	28,117	
Silurus glanis	2010s	0	0	0	0			

Notes: Due to uncertainties in identification, *Mimulus guttatus, Mimulus luteus* and *Mimulus guttatus x luteus* = *Mimulus guttatus/luteus* group, and *Physella acuta* and *Physella gyrina* = *Physella*. There were no records of *Mimulus ringens* or *Myriophyllum heterophyllum*.

Note that for early decades, representation of widespread taxa may not be complete due to lower recording effort.

[1] See Section 3.2 for definitions and methods of calculation.

AHULL = alpha hull; GR = ; MCP = minimum convex polygon; MCP INT = MCP intersection

Appendix B: Area of extent maps

For each species, the first map shows the 10km occurrence data, the second map shows the MCP (outlined by a red line) and its intersection with the land (green filled region) and third map shows the alpha hull and its intersection with the land (green filled region). The labels above each map give the total area of distinct 10km squares, the area of the MCP/England land intersection and the area of the alpha hull/England land intersection respectively.

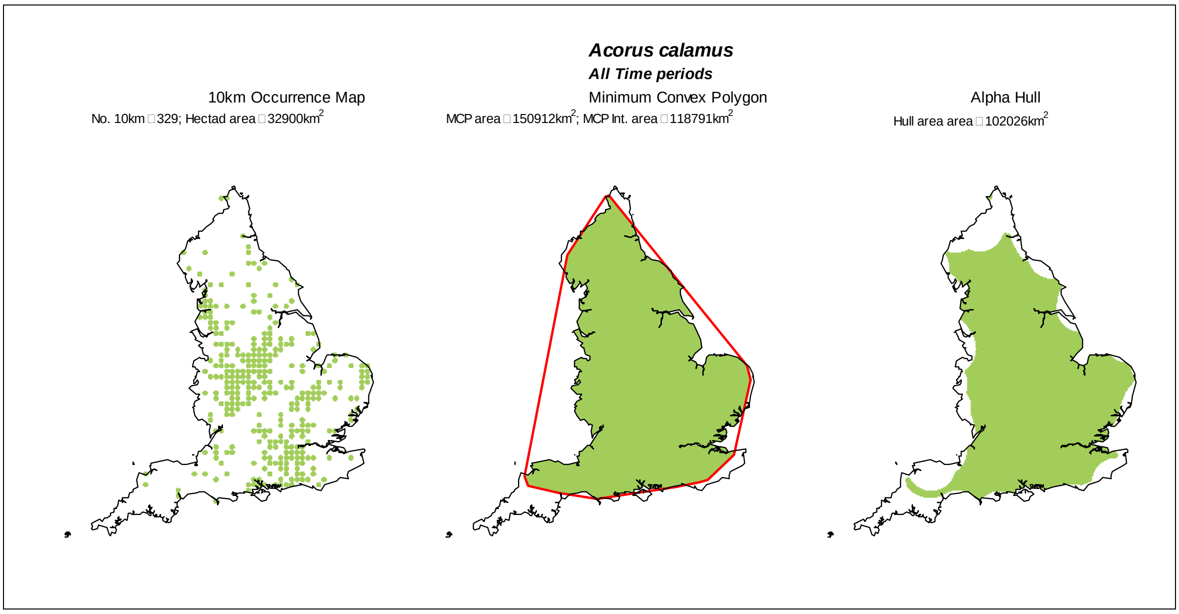


Figure B.1 Area of extent maps for *Acorus calamus* using all records

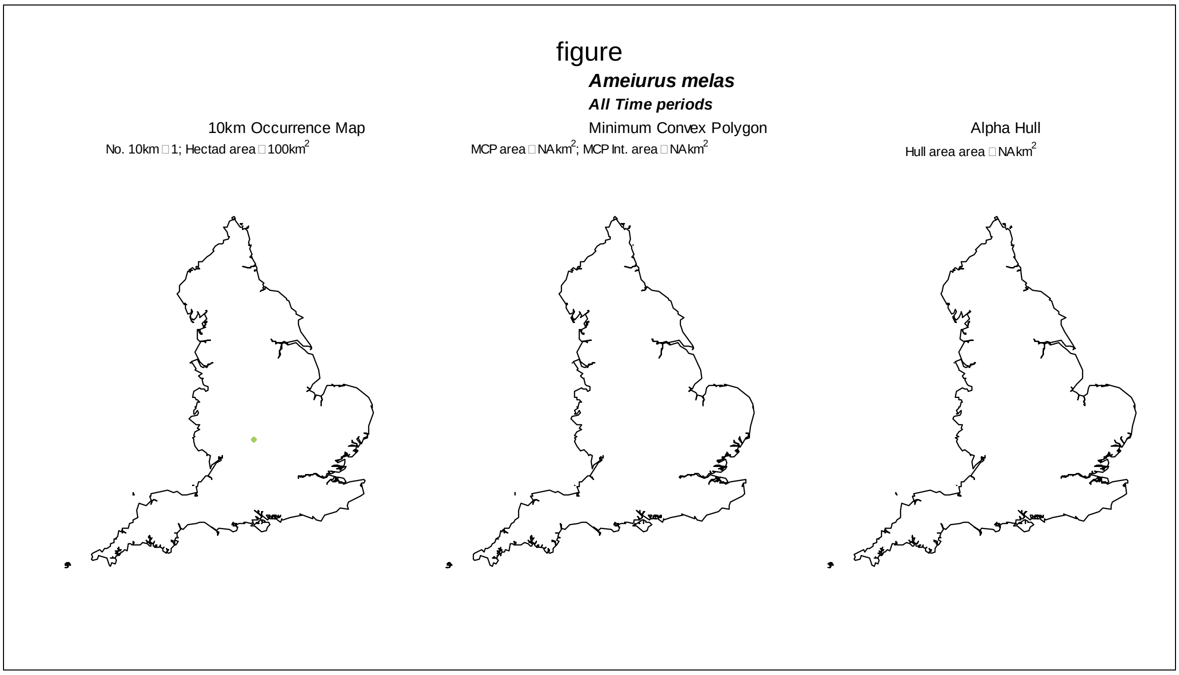


Figure B.2 Area of extent maps for *Ameiurus melas* using all records

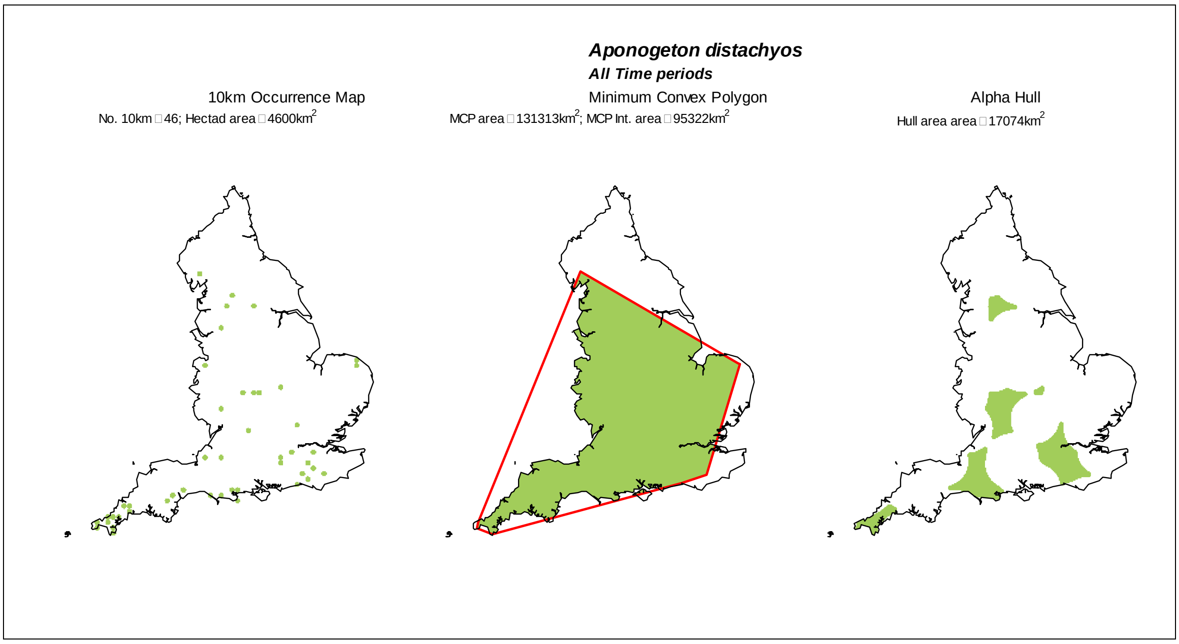


Figure B.3 Area of extent maps for *Aponogeton distachyos* using all records

Figure B.4 Area of extent maps for *Astacus astacus* using all records

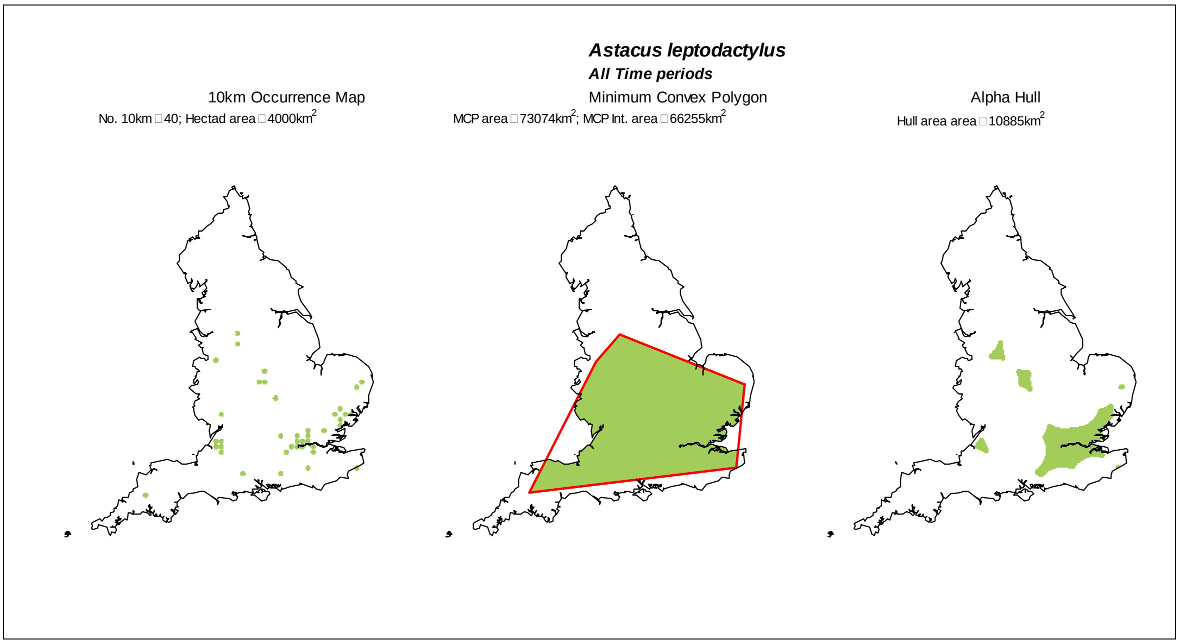


Figure B.5 Area of extent maps for *Astacus leptodactylus* using all records

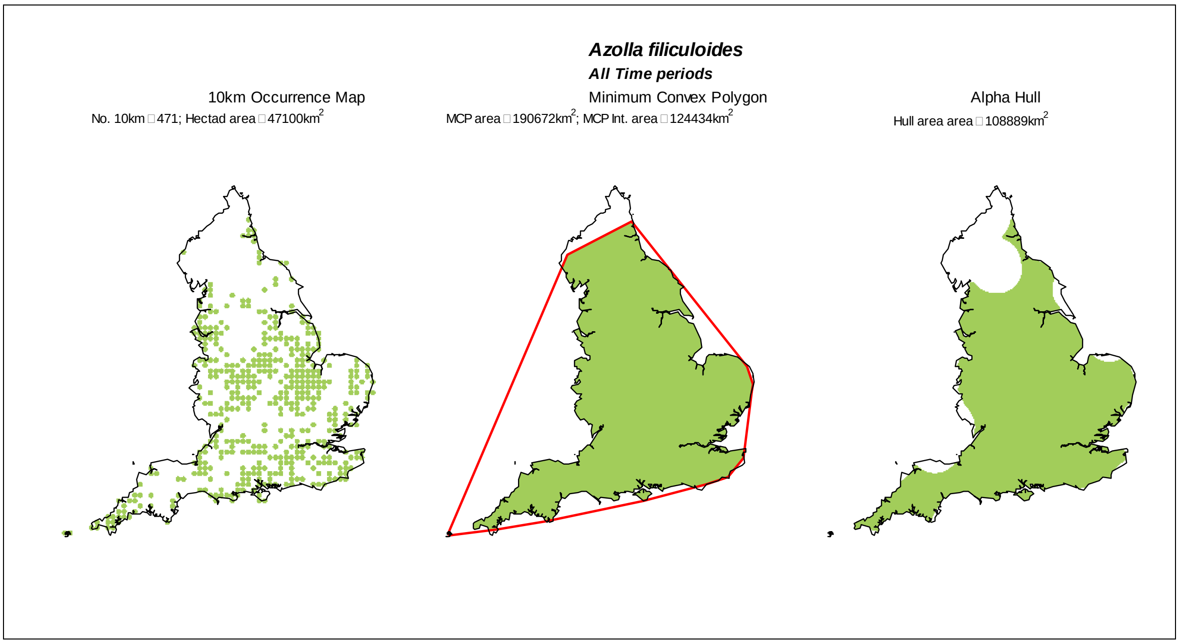


Figure B.6 Area of extent maps for *Azolla filiculoides* using all records

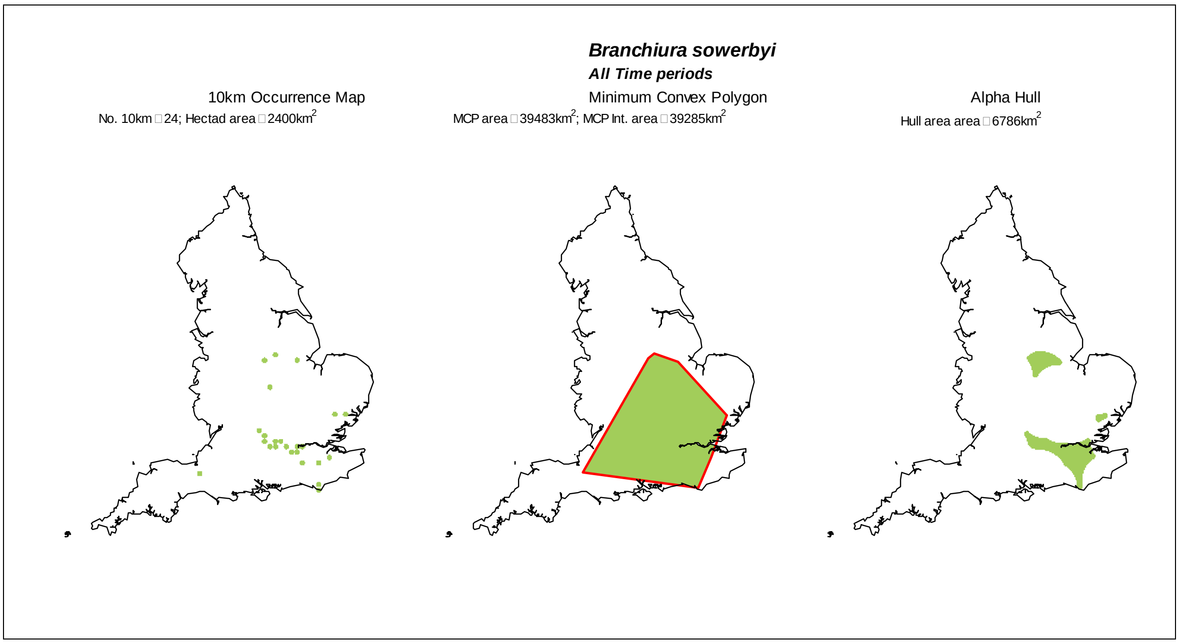


Figure B.7 Area of extent maps for *Branchiura sowerbyi* using all records

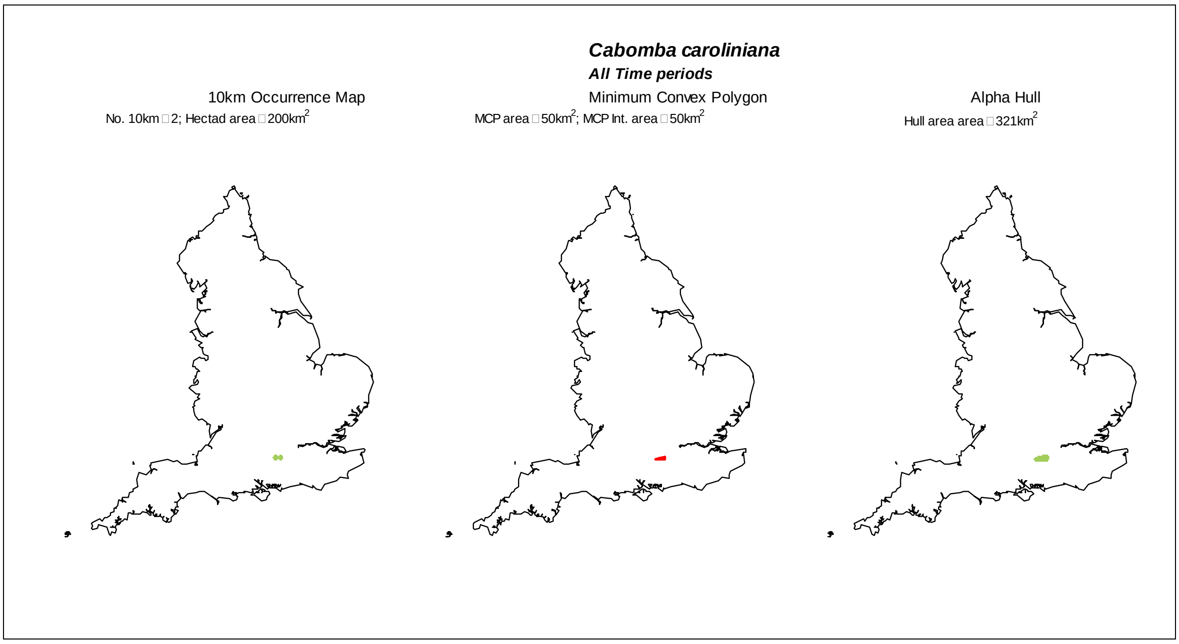


Figure B.8 Area of extent maps for *Cabomba caroliniana* using all records

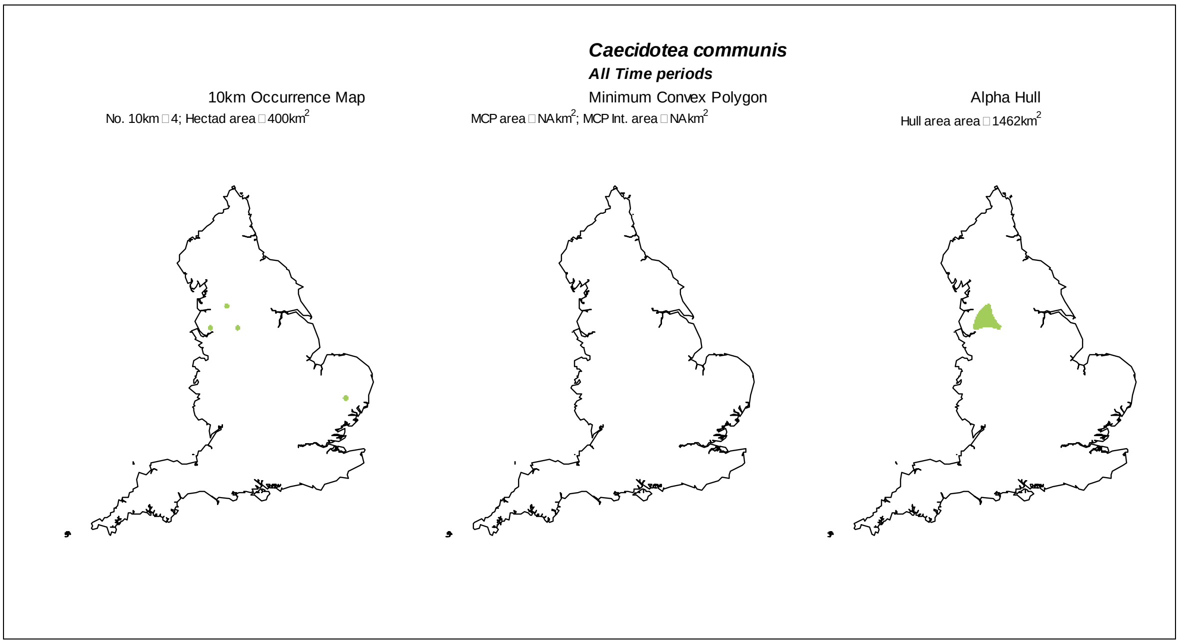


Figure B.9 Area of extent maps for *Caecidotea communis* using all records

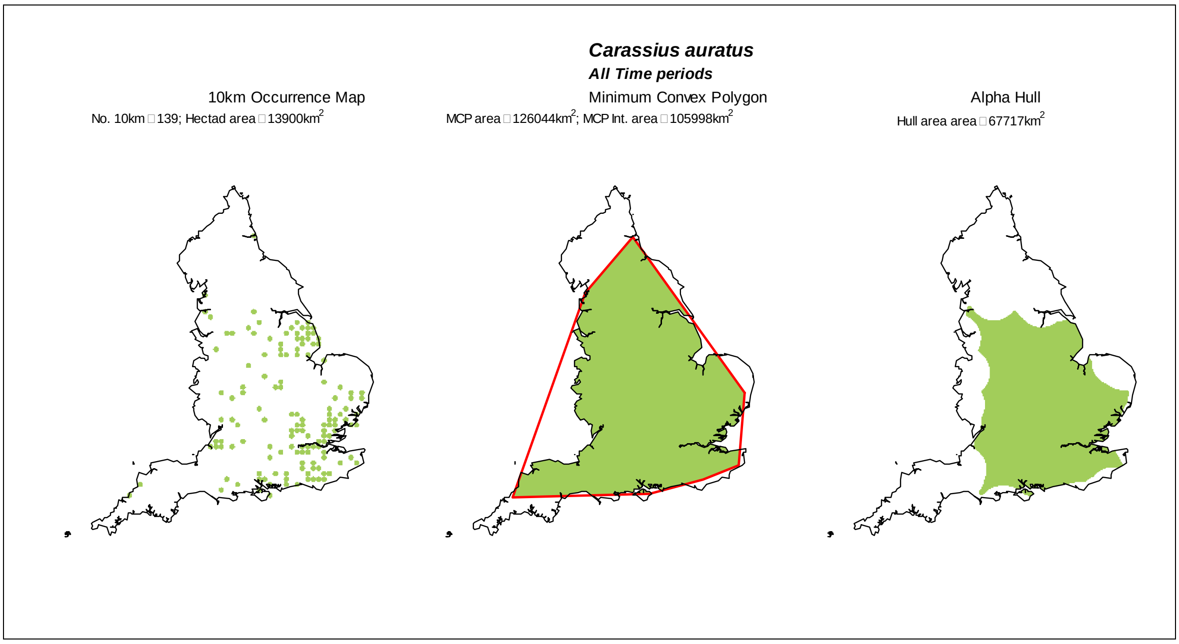


Figure B.10 Area of extent maps for *Carassius auratus* using all records

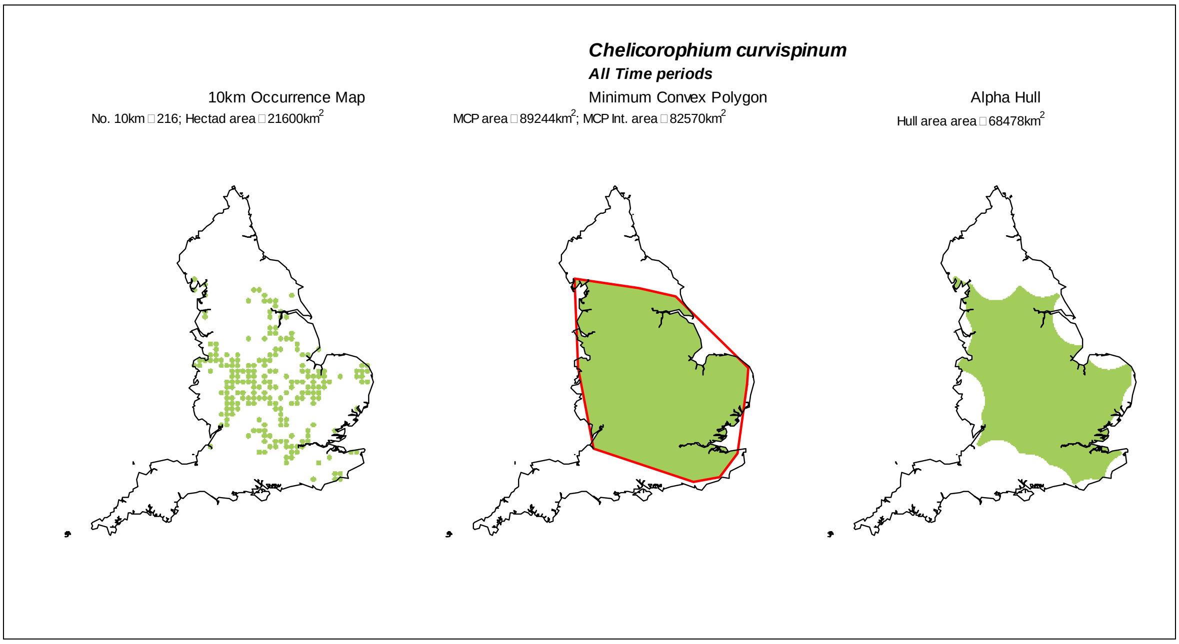


Figure B.11 Area of extent maps for *Chelicorophium curvispinum* using all records

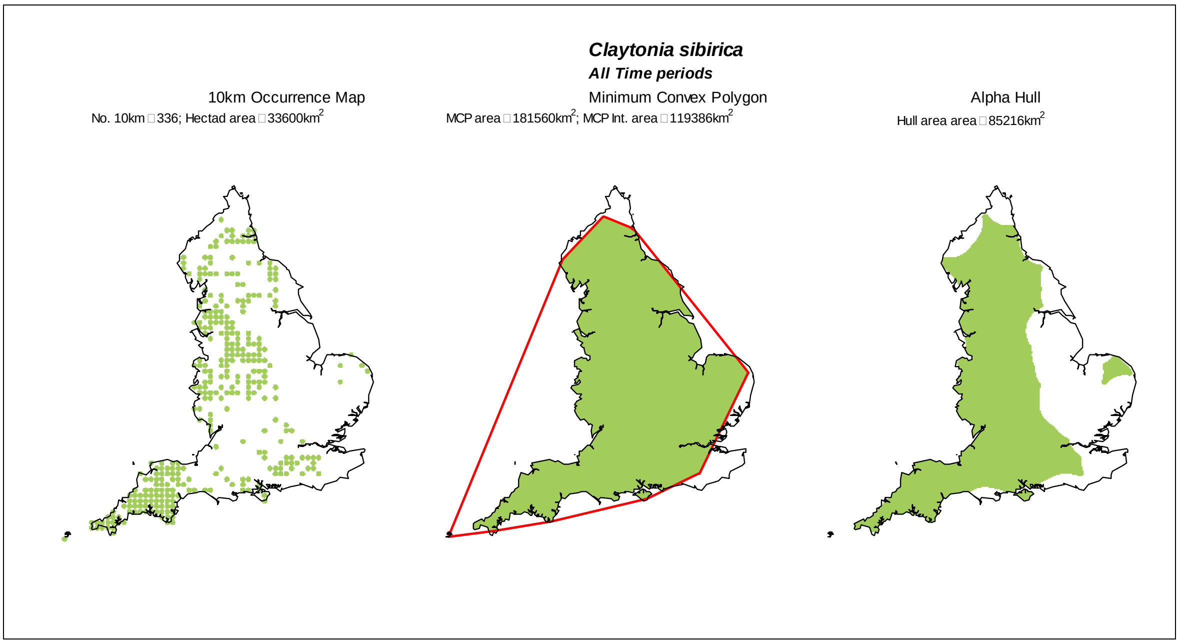


Figure B.12 Area of extent maps for *Claytonia sibirica* using all records

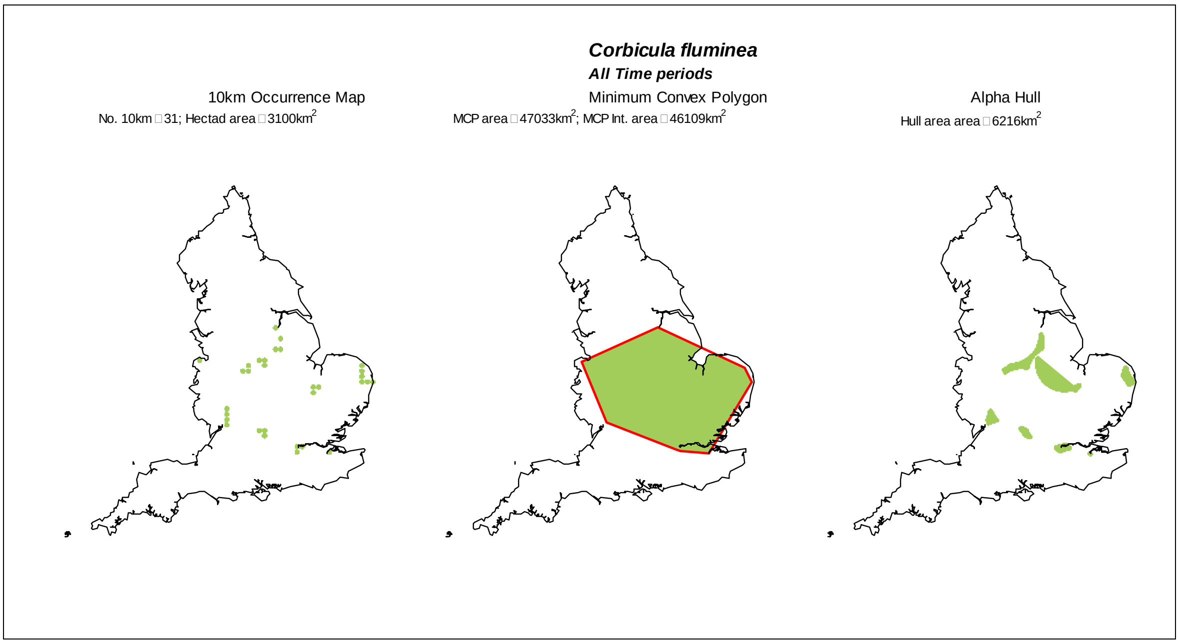


Figure B.13 Area of extent maps for *Corbicula fluminea* using all records

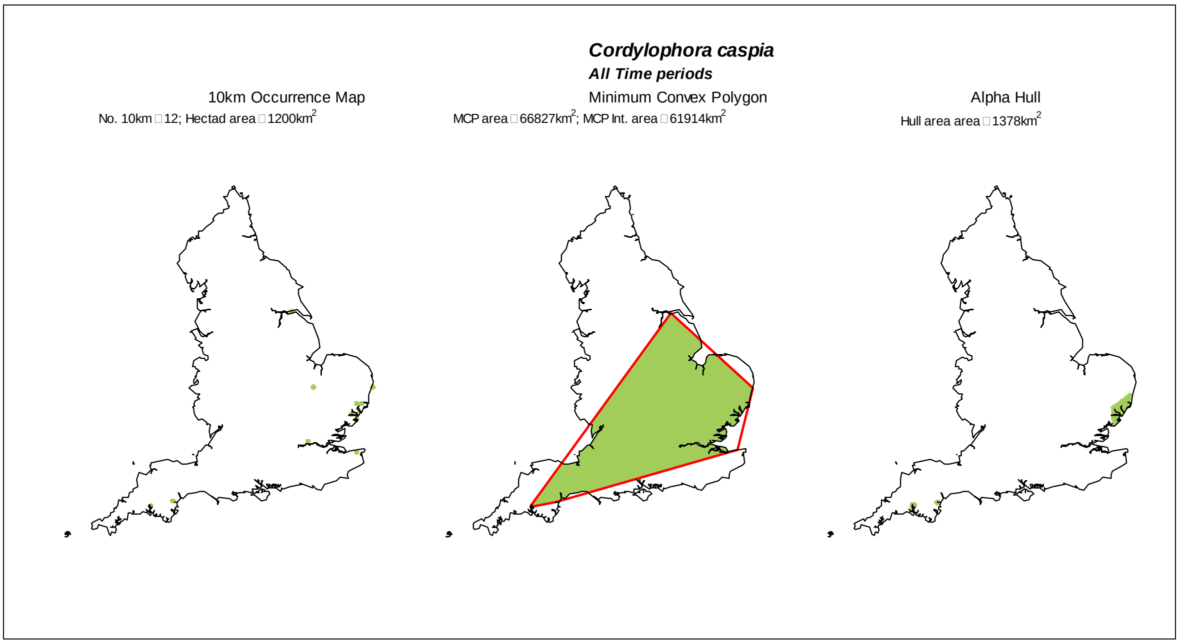


Figure B.14 Area of extent maps for *Cordylophora caspia* using all records

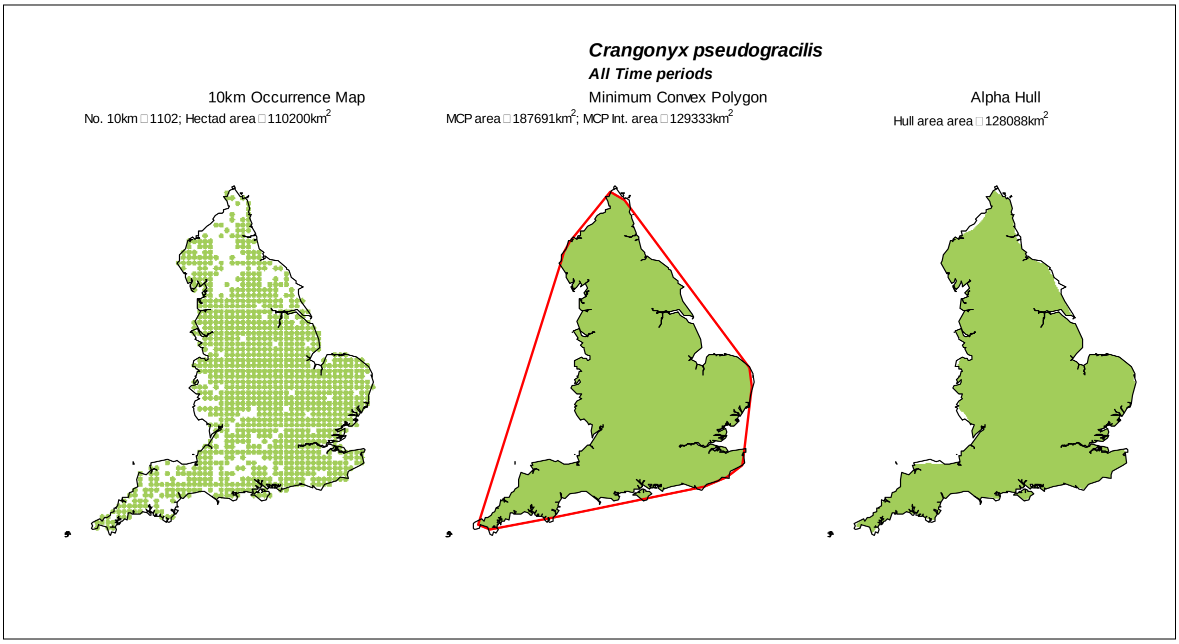


Figure B.15 Area of extent maps for *Crangonyx pseudogracilis* using all records

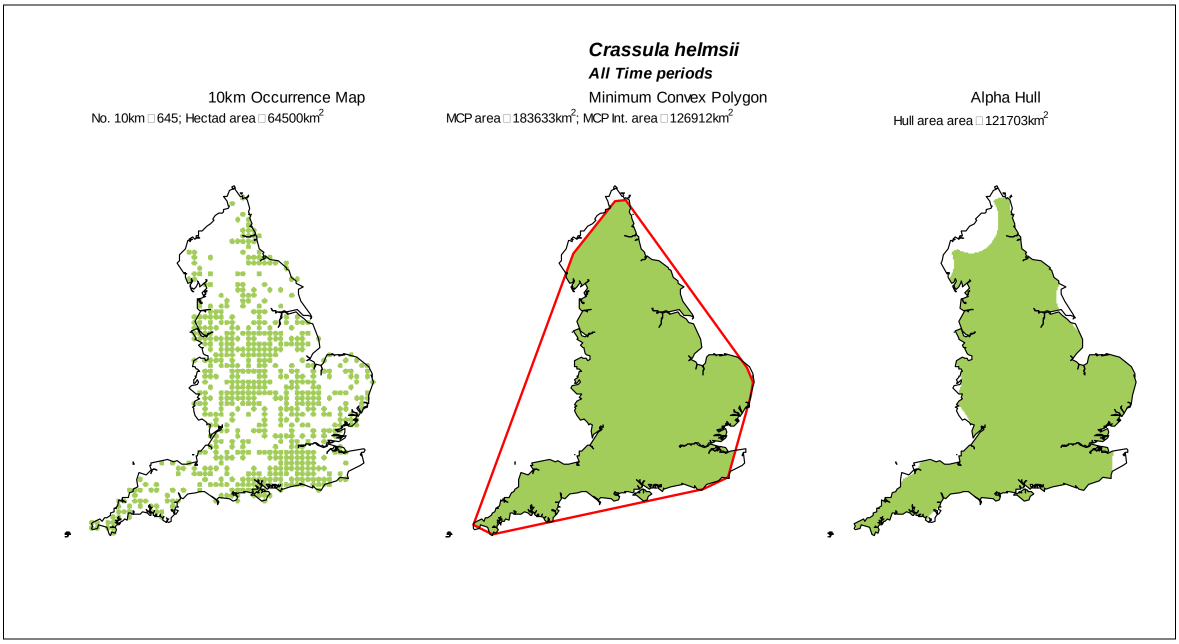


Figure B.16 Area of extent maps for *Crassula helmsii* using all records

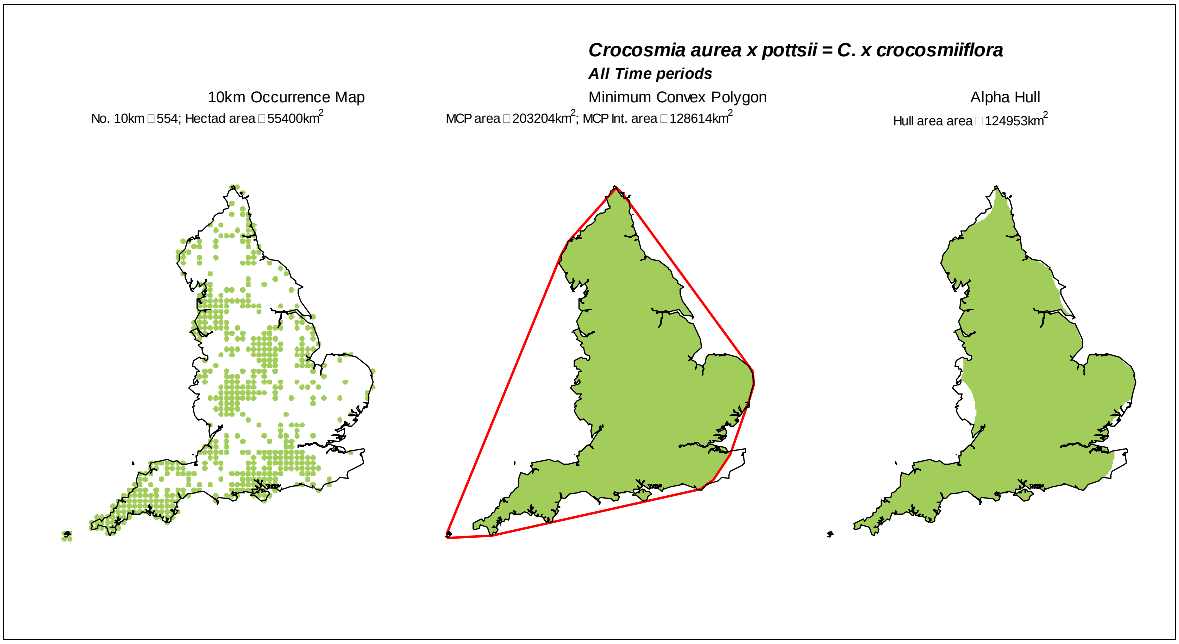


Figure B.17 Area of extent maps for *Crocosmia aurea x pottsii (C. x crocosmiiflora)* using all records

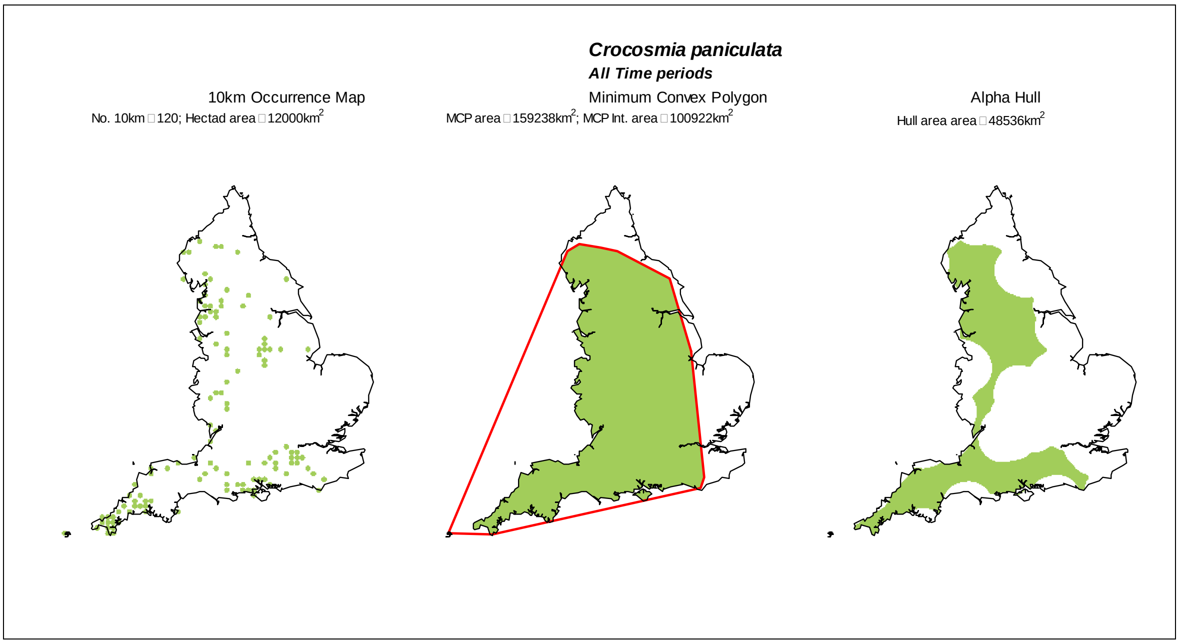


Figure B.18 Area of extent maps for *Crocosmia paniculata* using all records

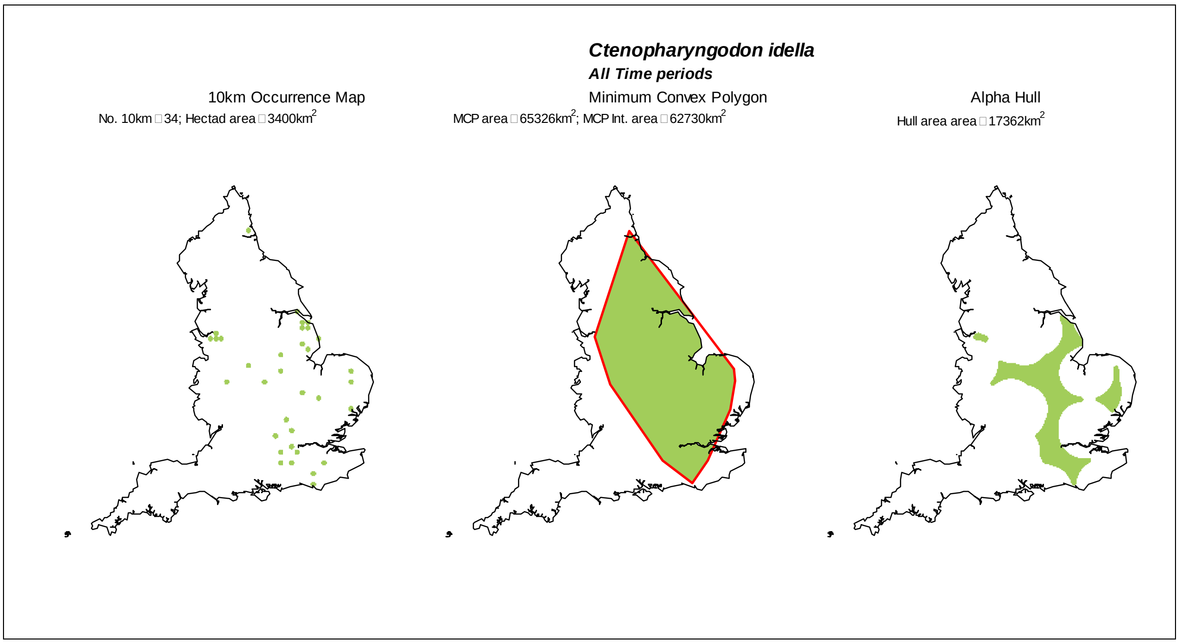


Figure B.19 Area of extent maps for *Ctenopharyngodon idella* using all records

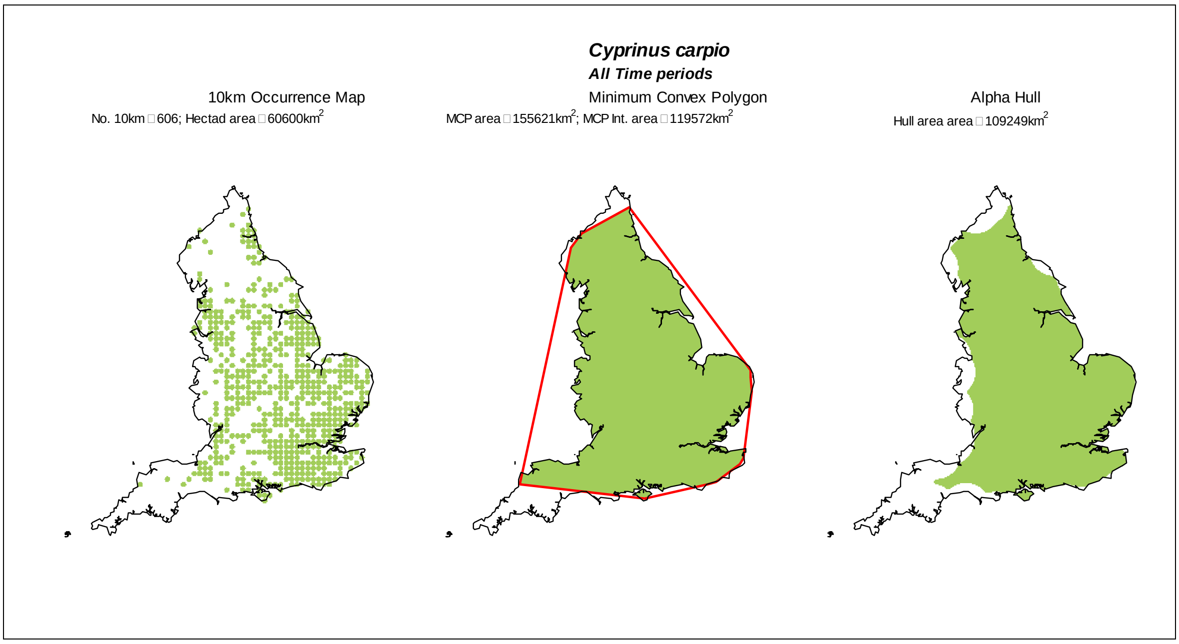


Figure B.20 Area of extent maps for *Cyprinus carpio* using all records

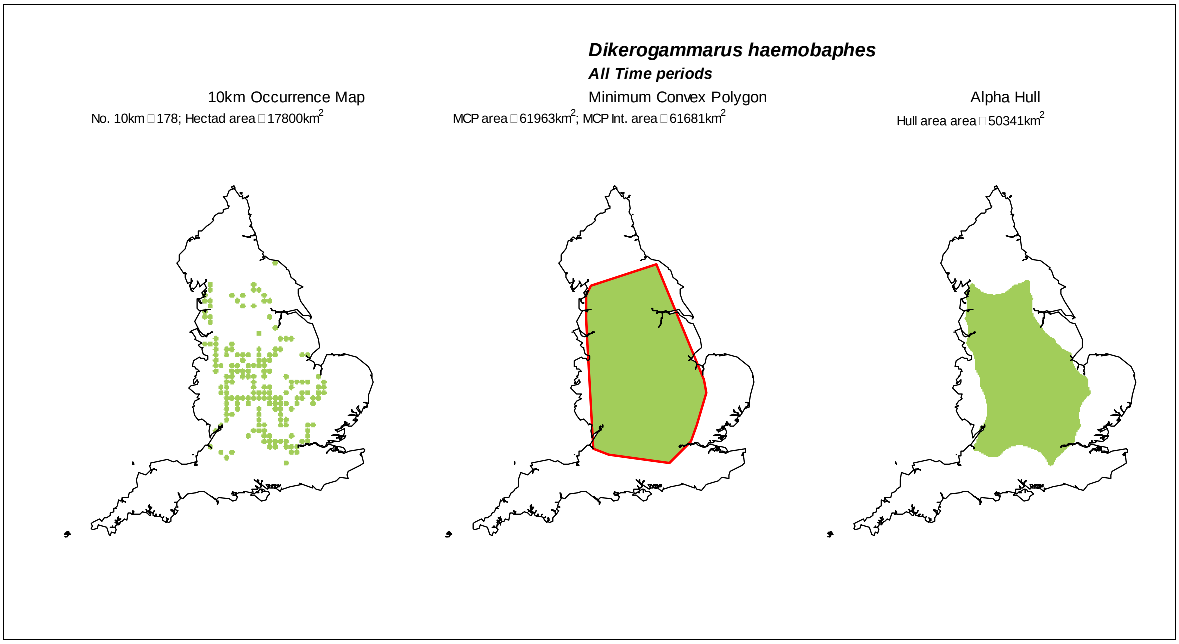


Figure B.21 Area of extent maps for *Dikerogammarus haemobaphes* using all records

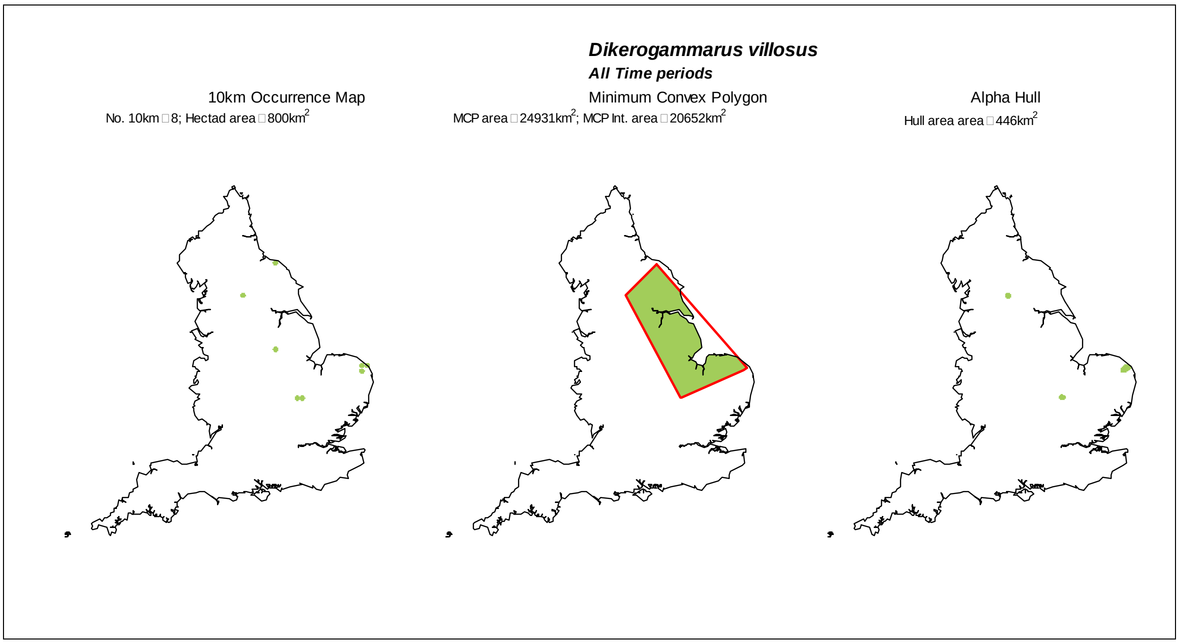


Figure B.22 Area of extent maps for *Dikerogammarus villosus* using all records

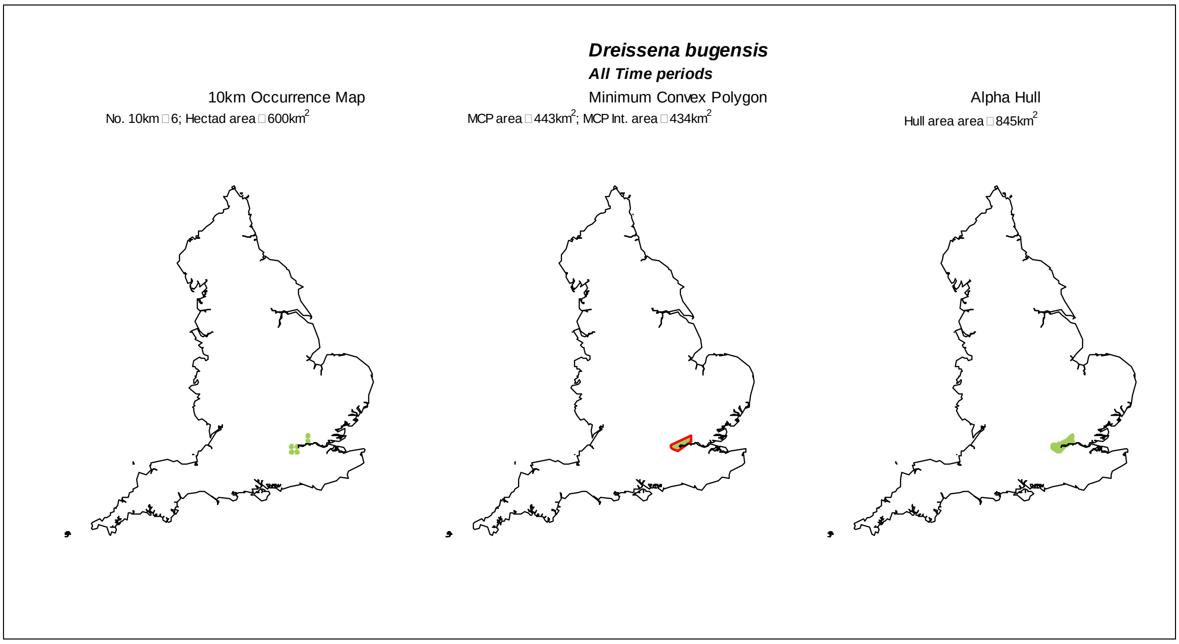


Figure B.23 Area of extent maps for *Dreissena bugensis* using all records

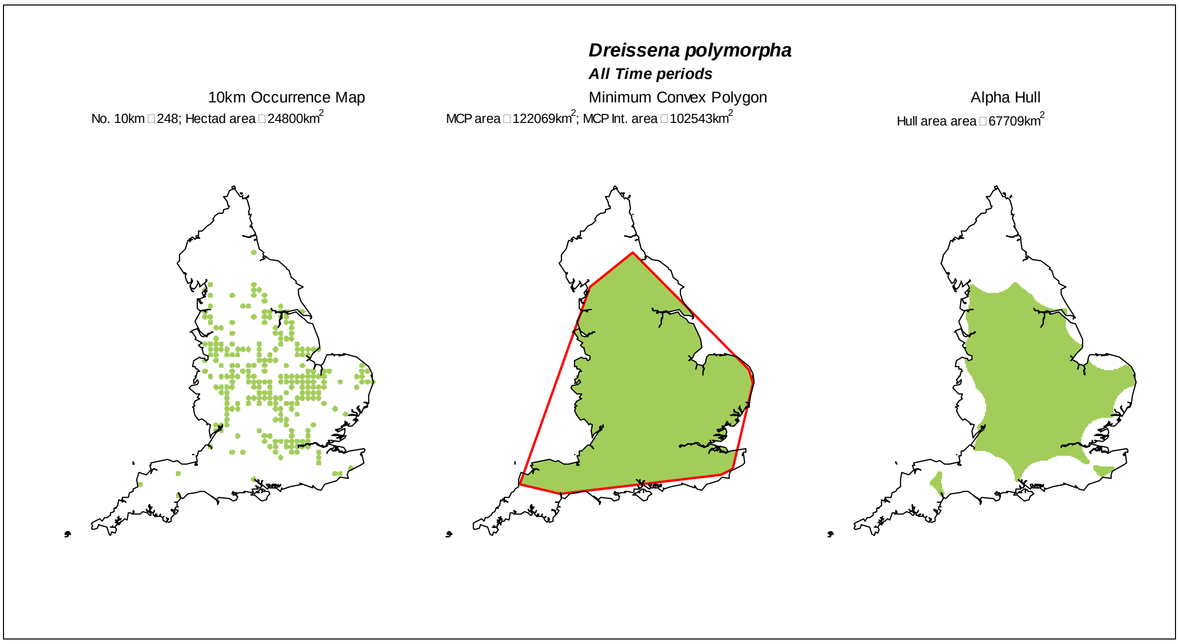


Figure B.24 Area of extent maps for *Dreissena polymorpha* using all records

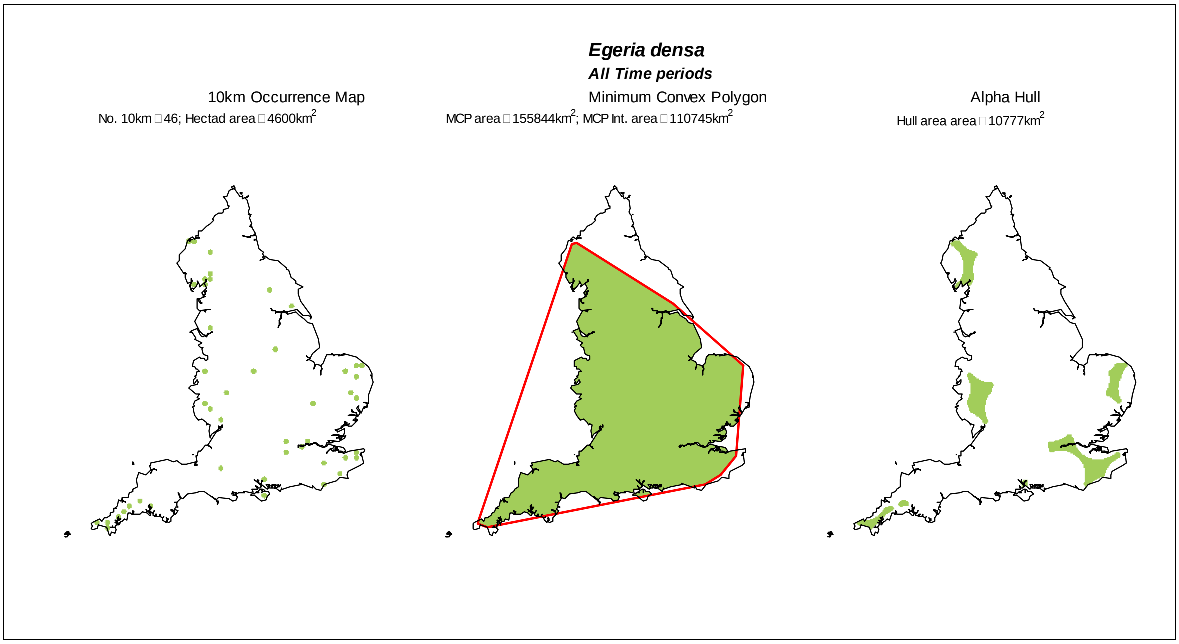


Figure B.25 Area of extent maps for *Egeria densa* using all records

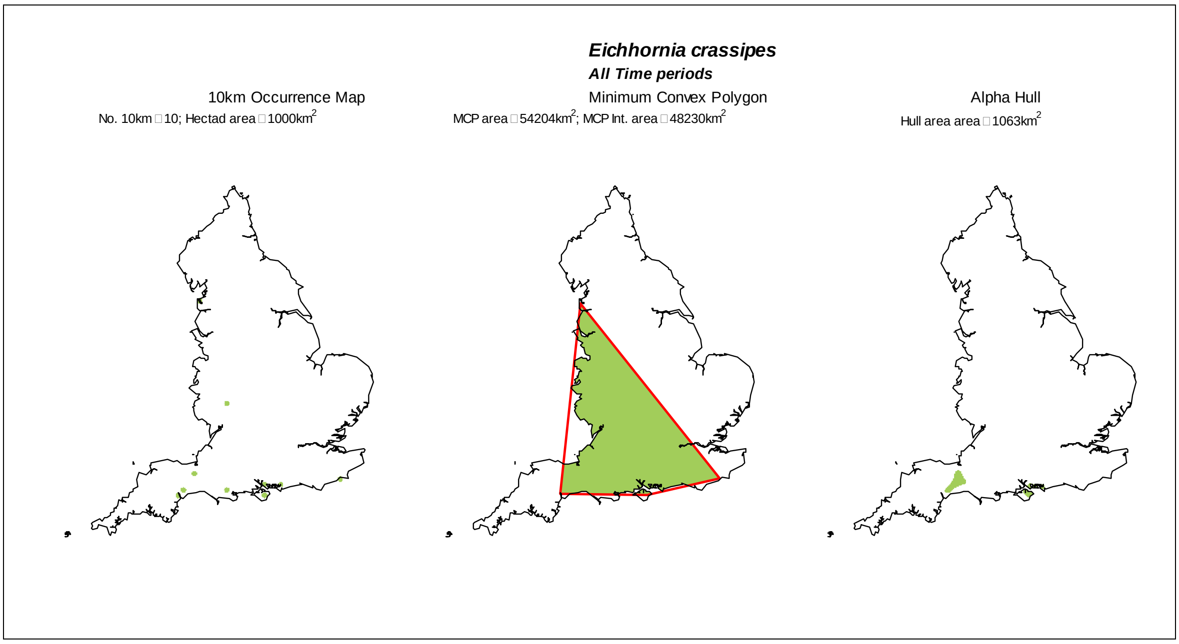


Figure B.26 Area of extent maps for *Eichhornia crassipes* using all records

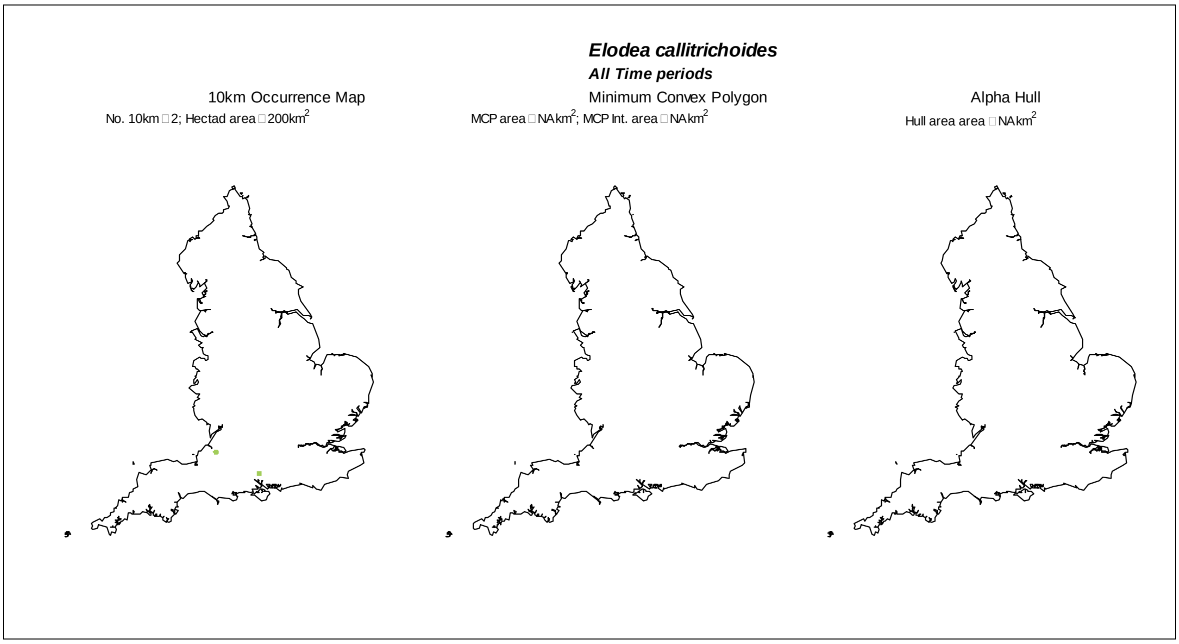


Figure B.27 Area of extent maps for *Elodea callitrichoides* using all records

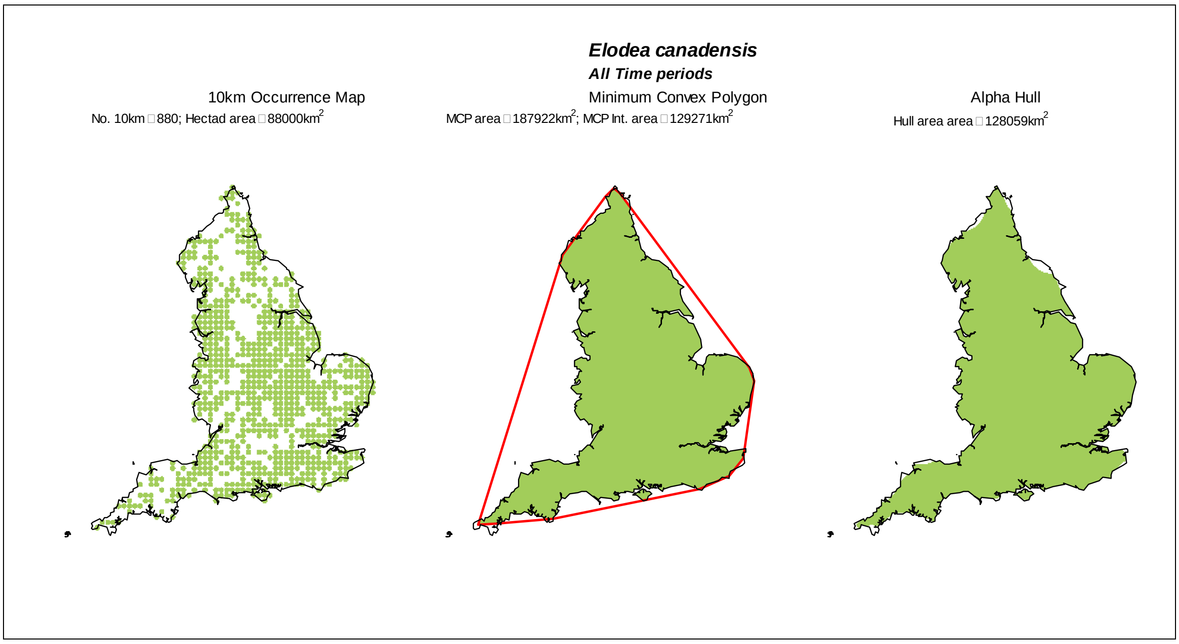


Figure B.28 Area of extent maps for *Elodea canadensis* using all records

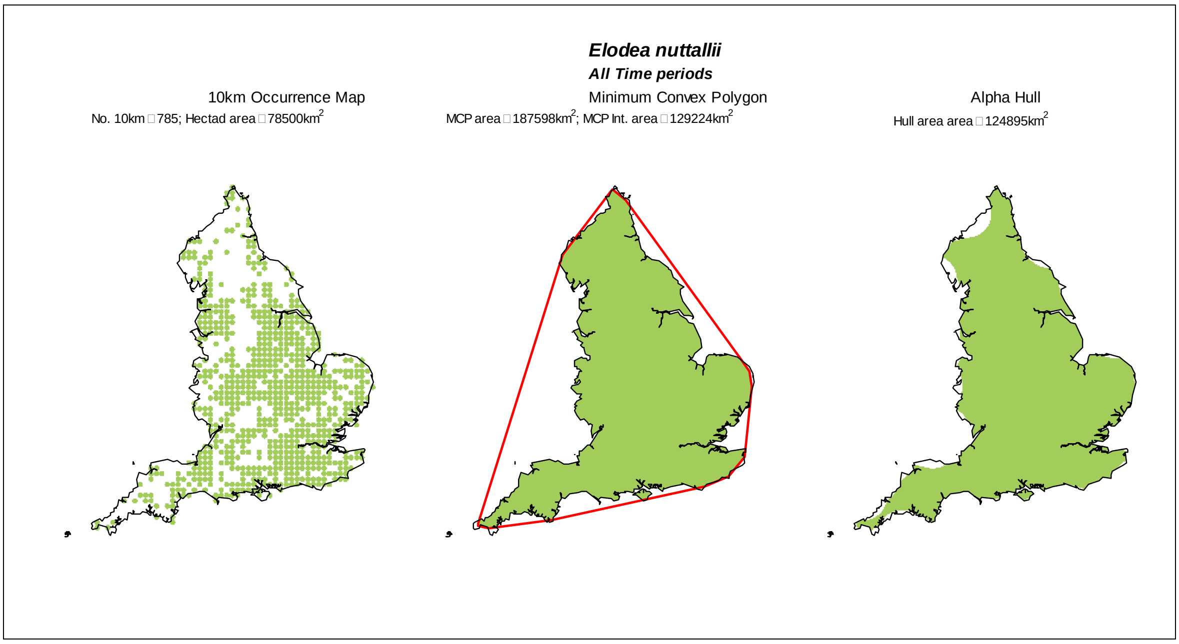


Figure B.29 Area of extent maps for *Elodea nuttallii* using all records

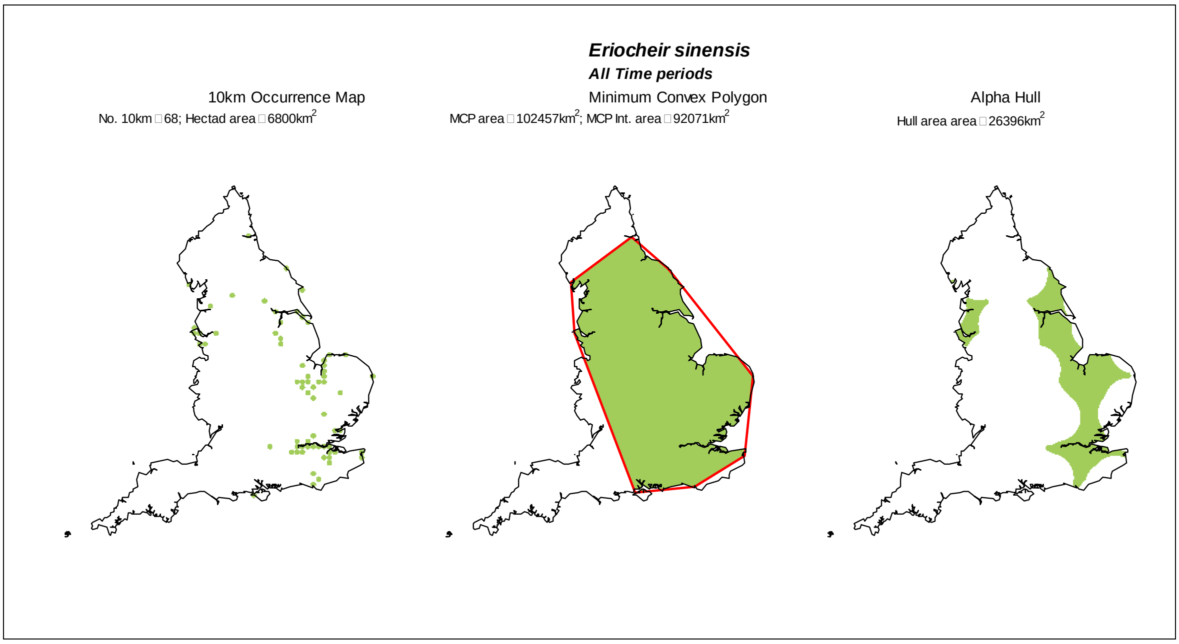


Figure B.30 Area of extent maps for *Eriocheir sinensis* using all records

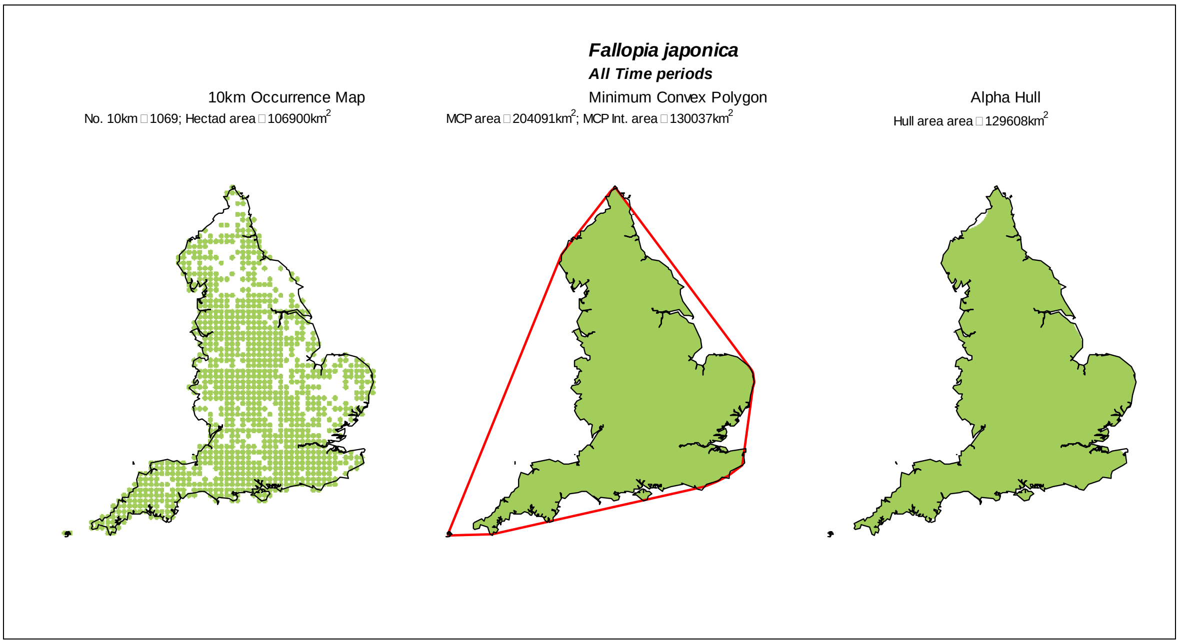


Figure B.31 Area of extent maps for *Fallopia japonica* using all records

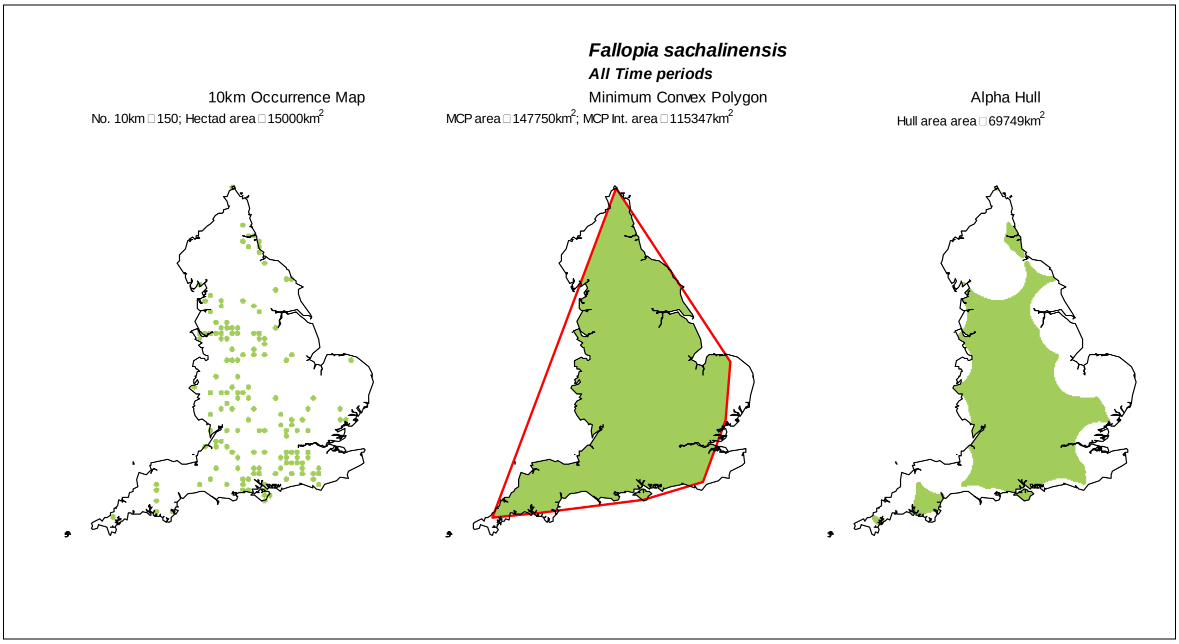


Figure B.32 Area of extent maps for *Fallopia sachalinensis* using all records

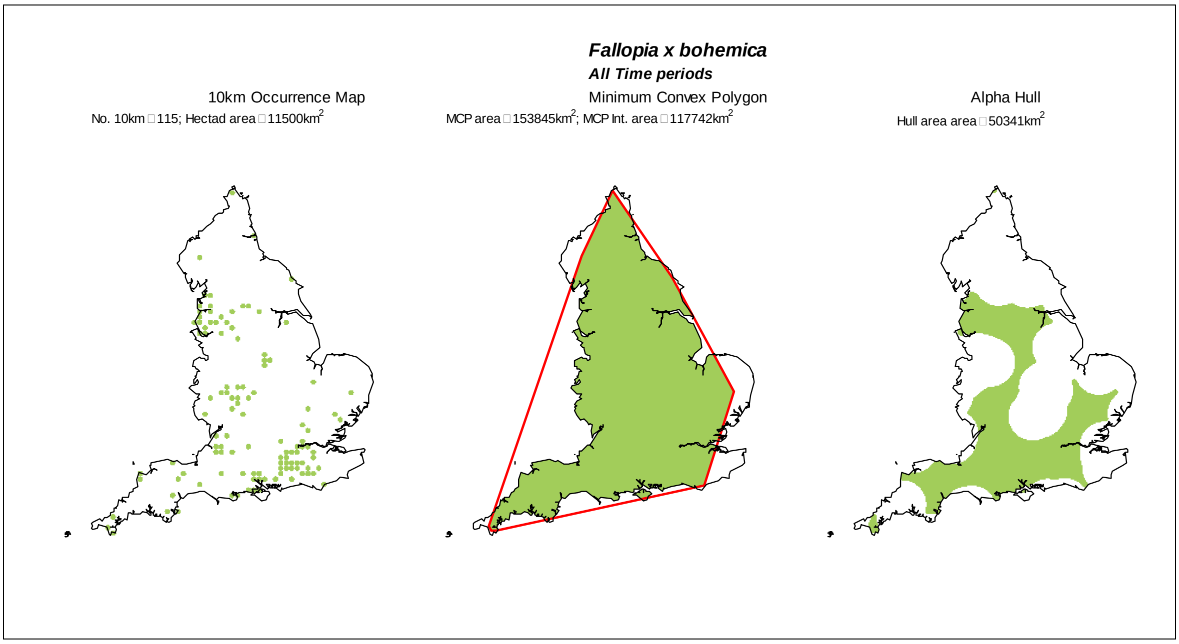


Figure B.33 Area of extent maps for *Fallopia x bohemica* using all records

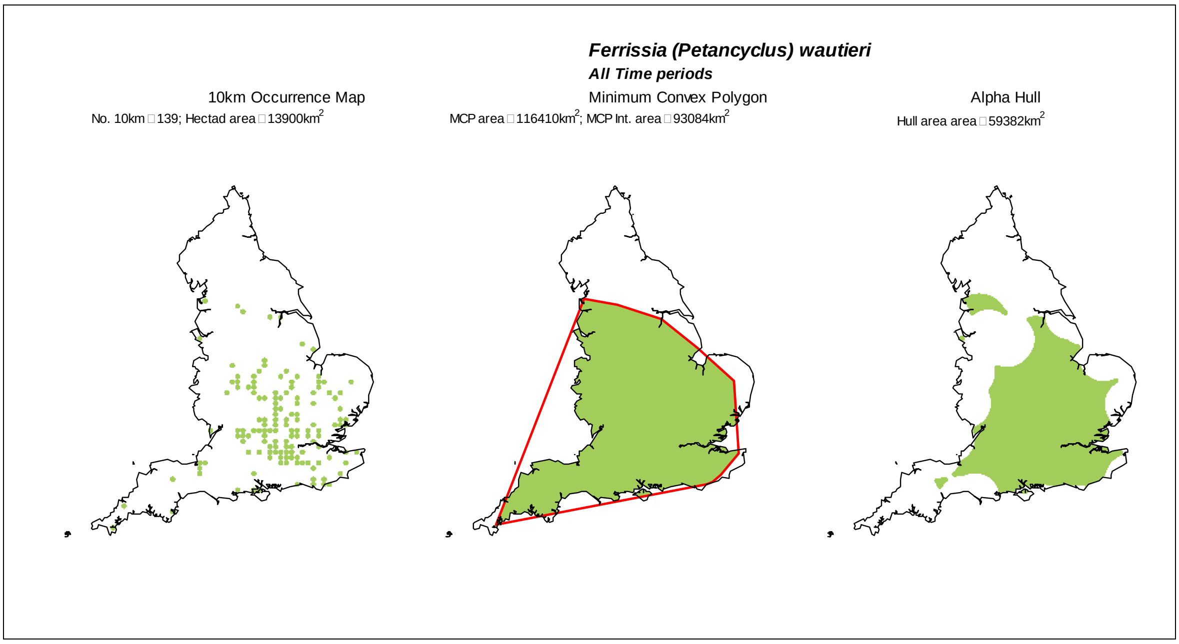


Figure B.34 Area of extent maps for *Ferrissia (Petancyclus) wautieri* using all records

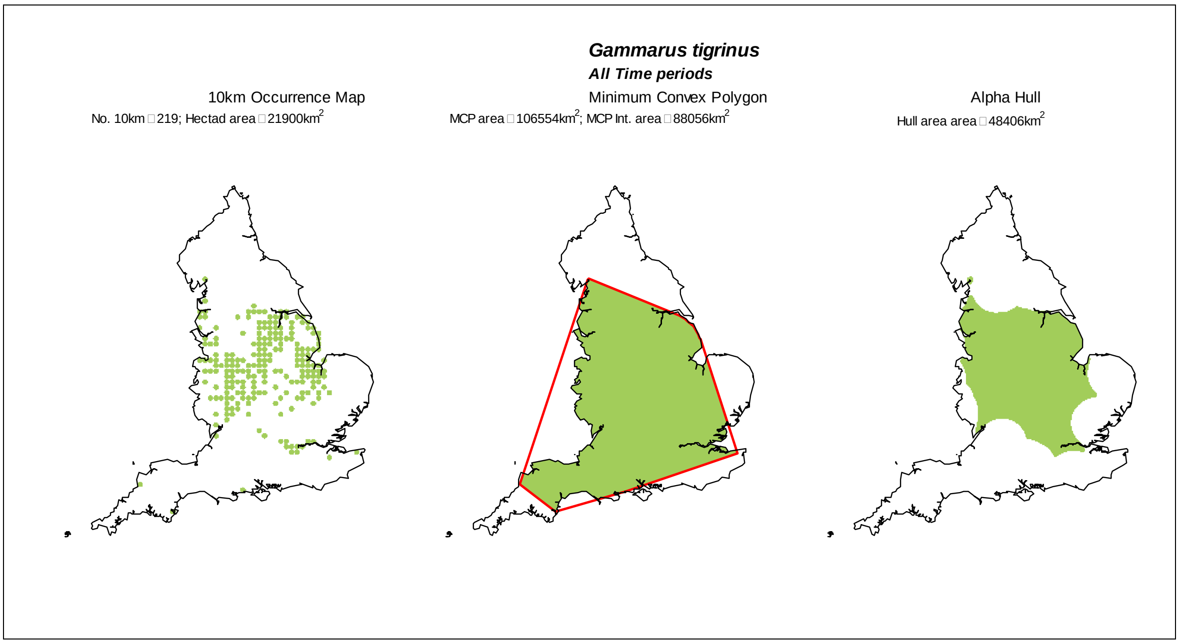


Figure B.35 Area of extent maps for *Gammarus tigrinus* using all records

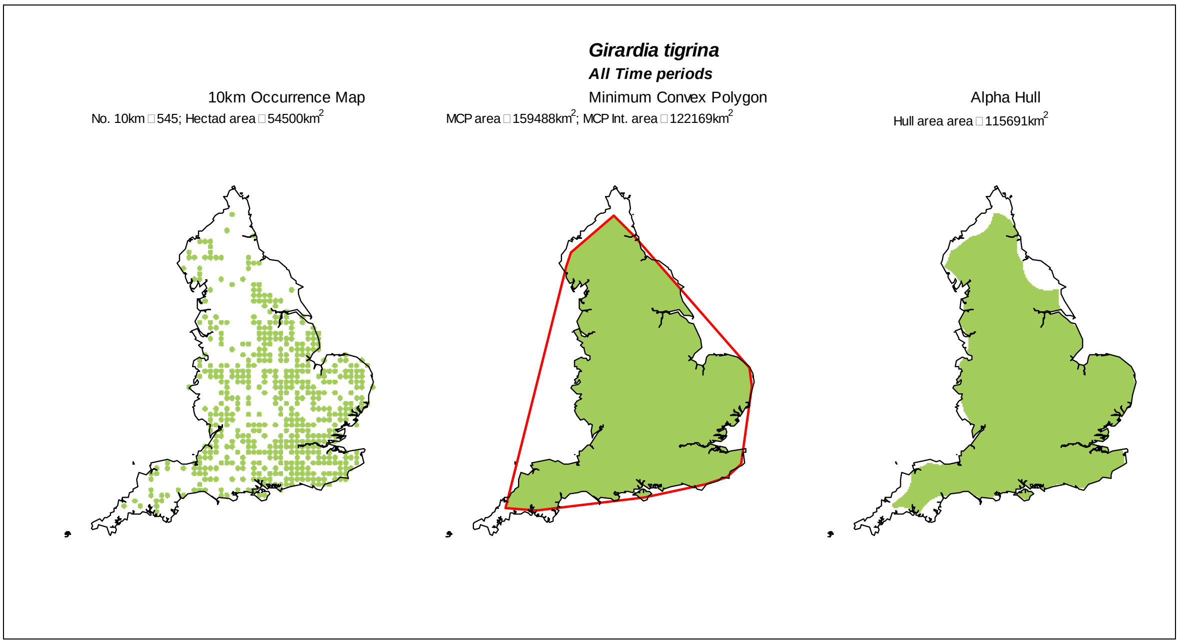


Figure B.36 Area of extent maps for *Girardia tigrina* using all records

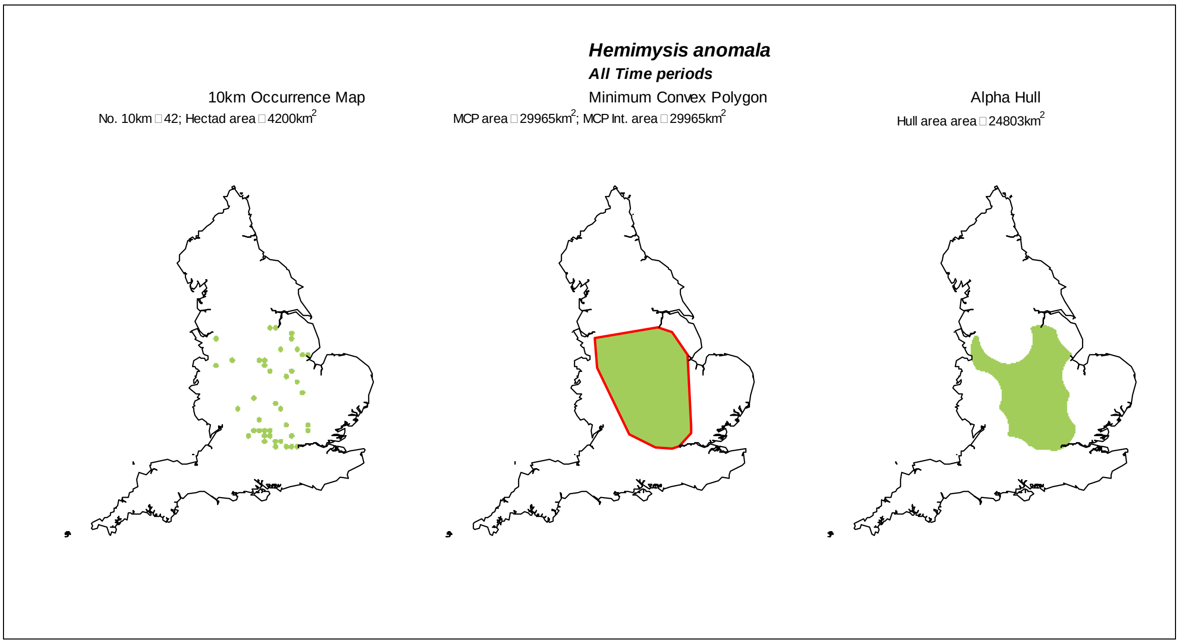


Figure B.37 Area of extent maps for *Hemimysis anomala* using all records

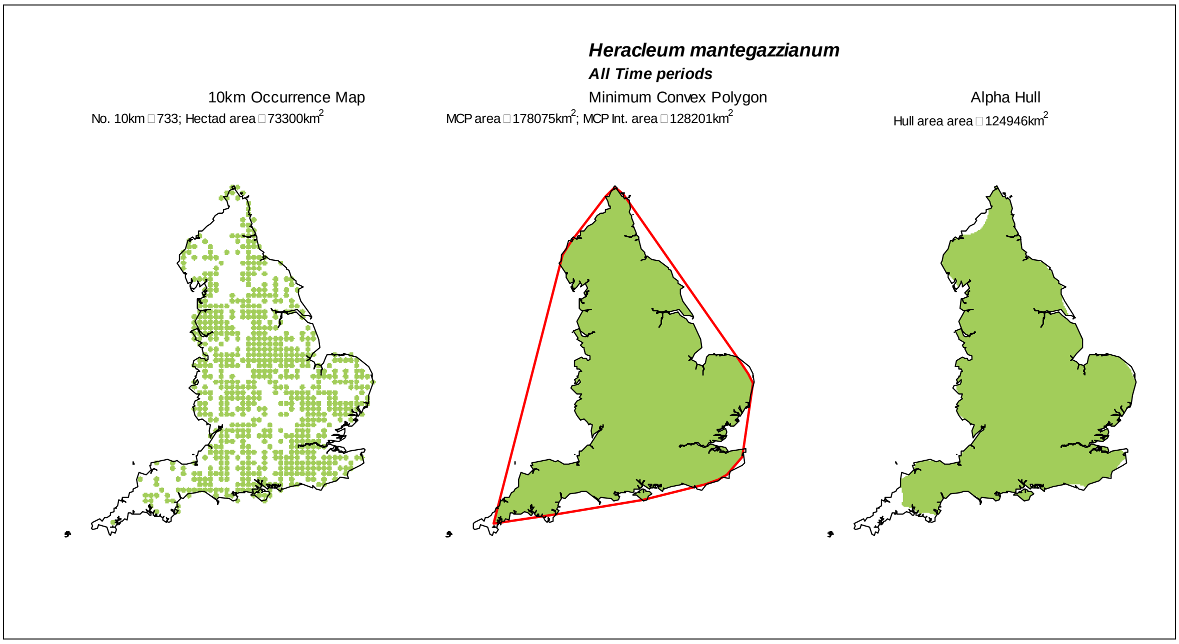


Figure B.38 Area of extent maps for *Heracleum mantegazzianum* using all records

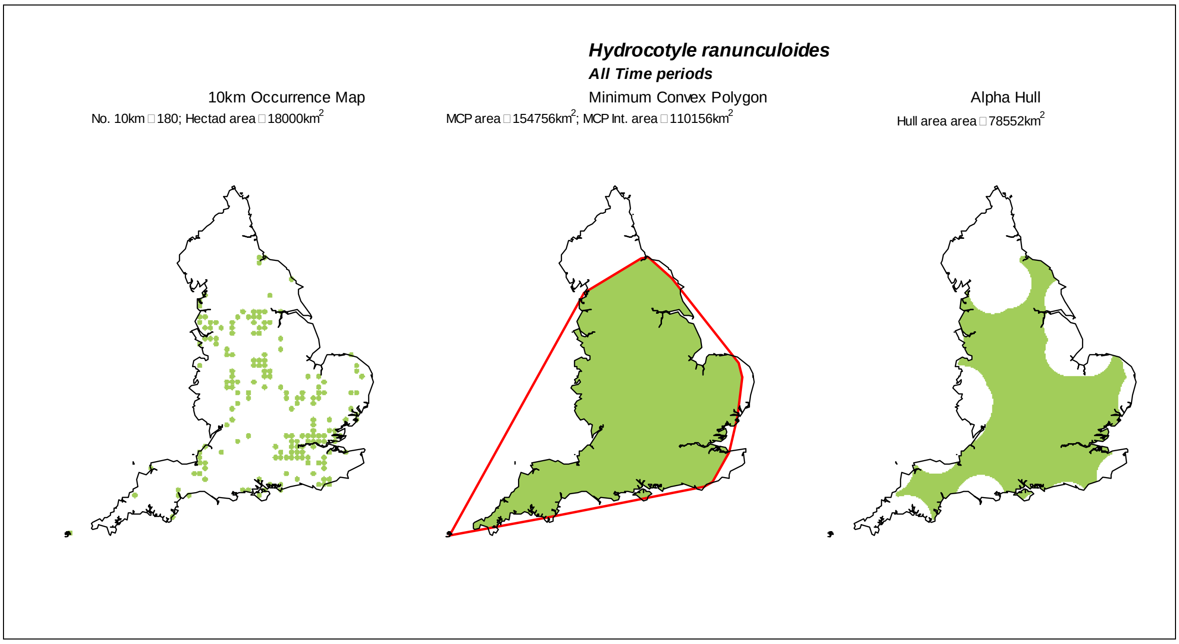


Figure B.39 Area of extent maps for *Hydrocotyle ranunculoides* using all records

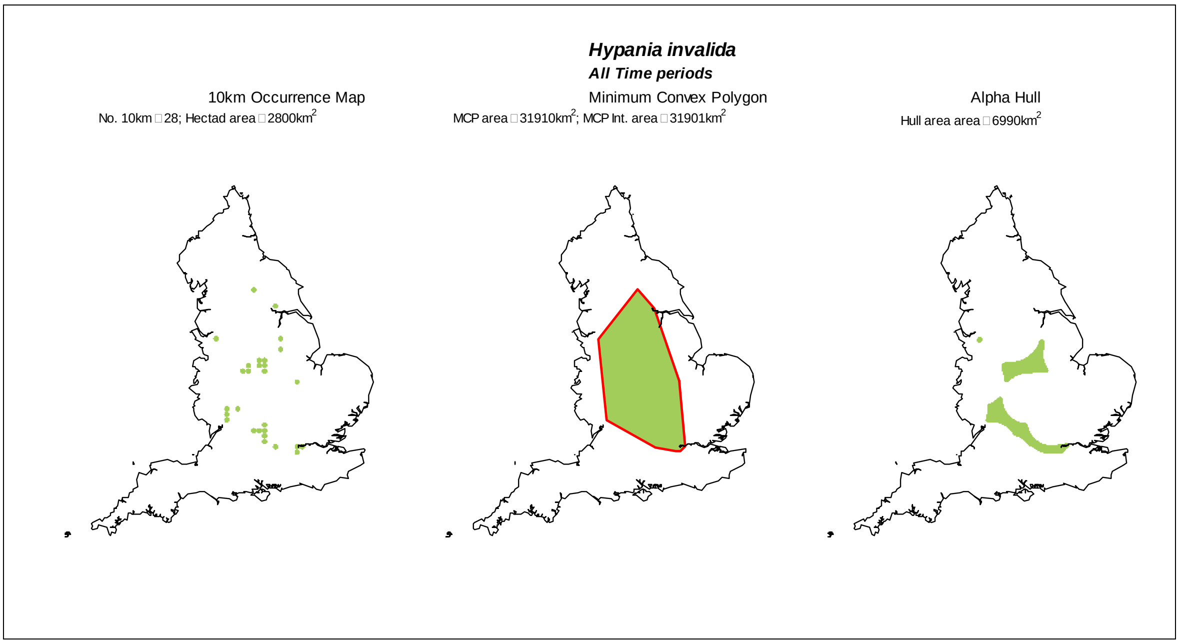


Figure B.40 Area of extent maps for *Hypania invalida* using all records

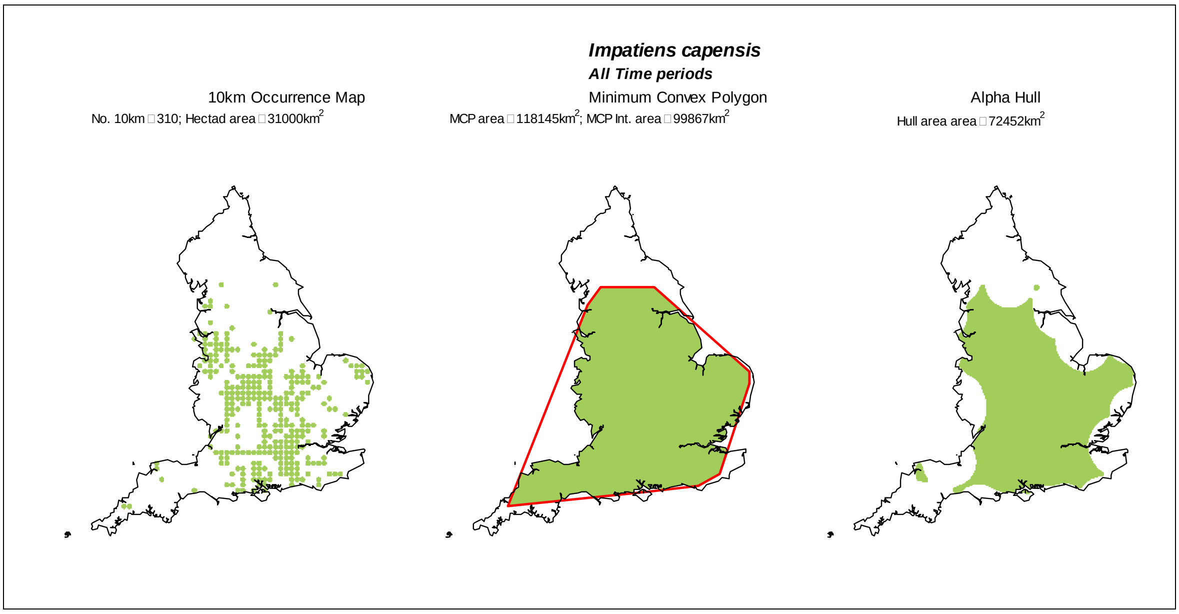


Figure B.41 Area of extent maps for *Impatiens capensis* using all records

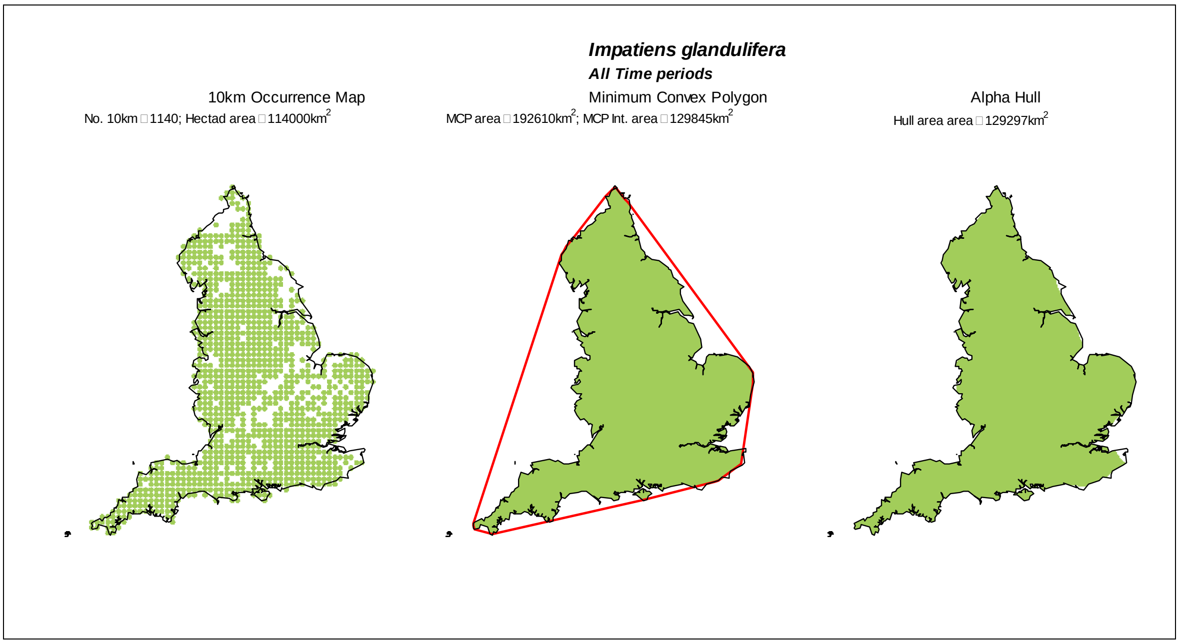


Figure B.42 Area of extent maps for *Impatiens glandulifera* using all records

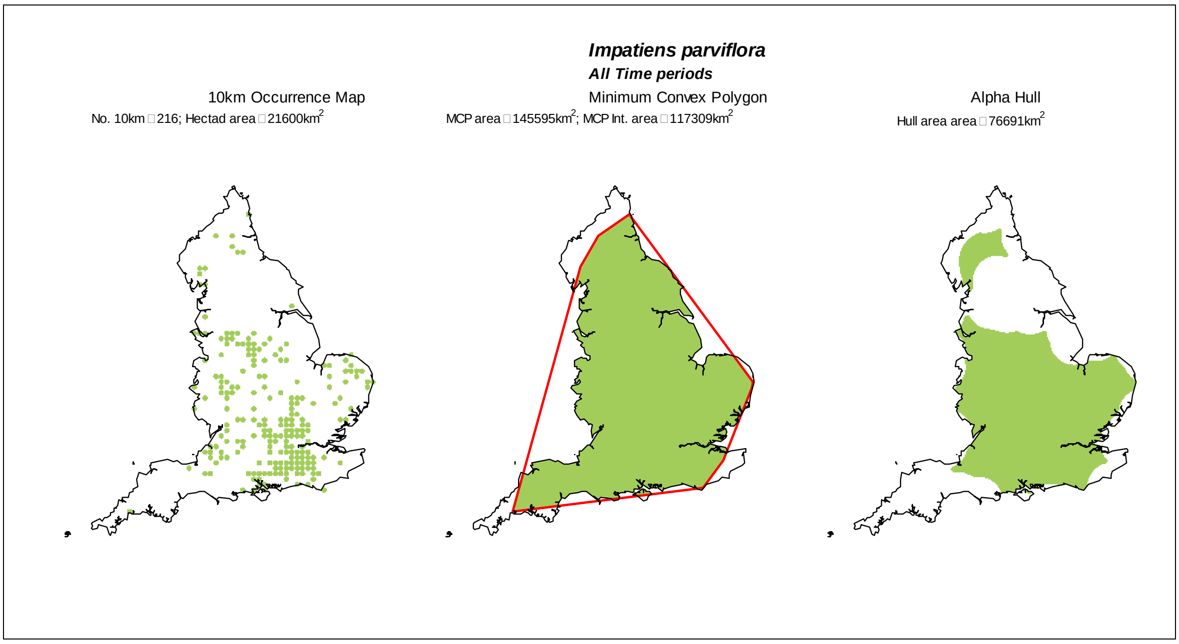


Figure B.43 Area of extent maps for *Impatiens parviflora* using all records

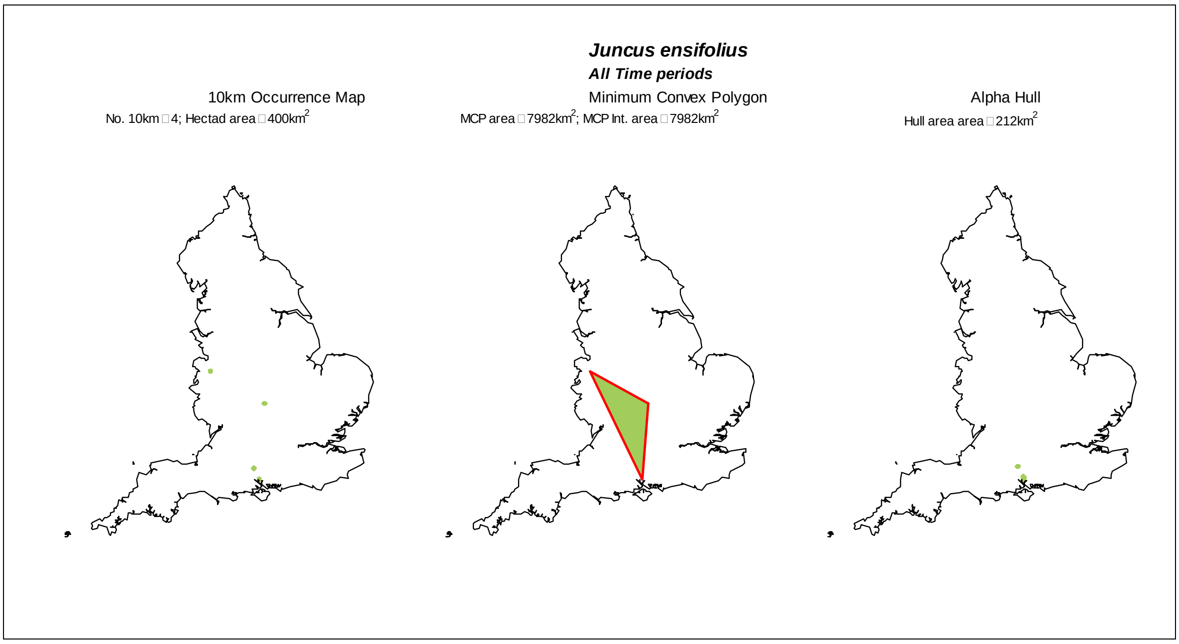


Figure B.44 Area of extent maps for *Juncus ensifolius* using all records

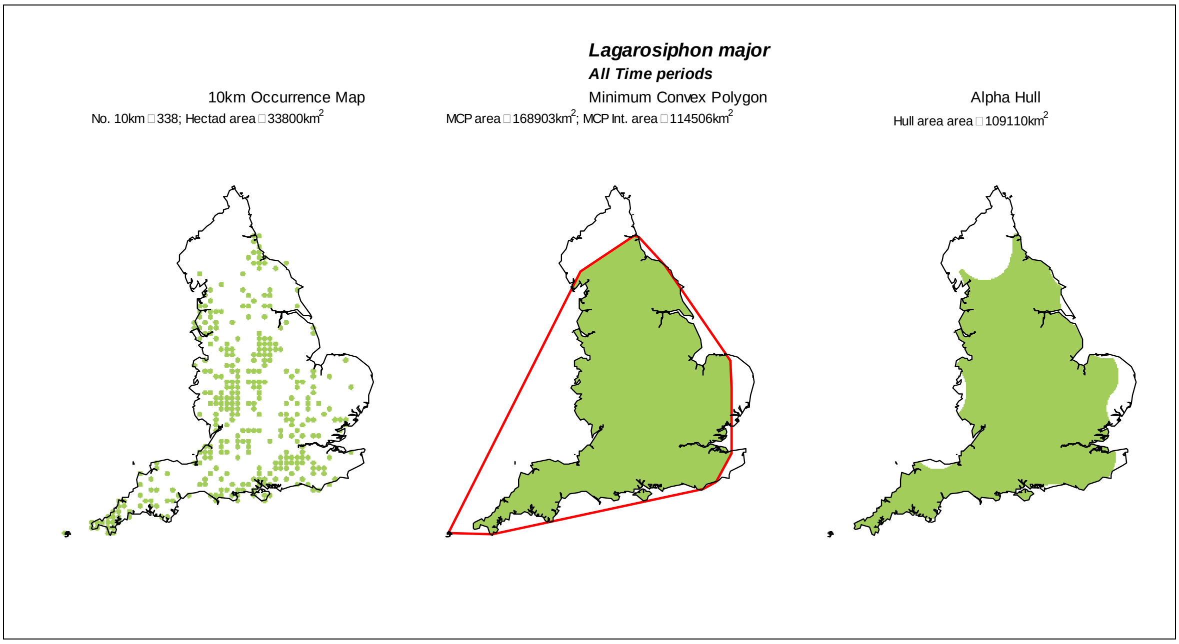


Figure B.45 Area of extent maps for *Lagarosiphon major* using all records

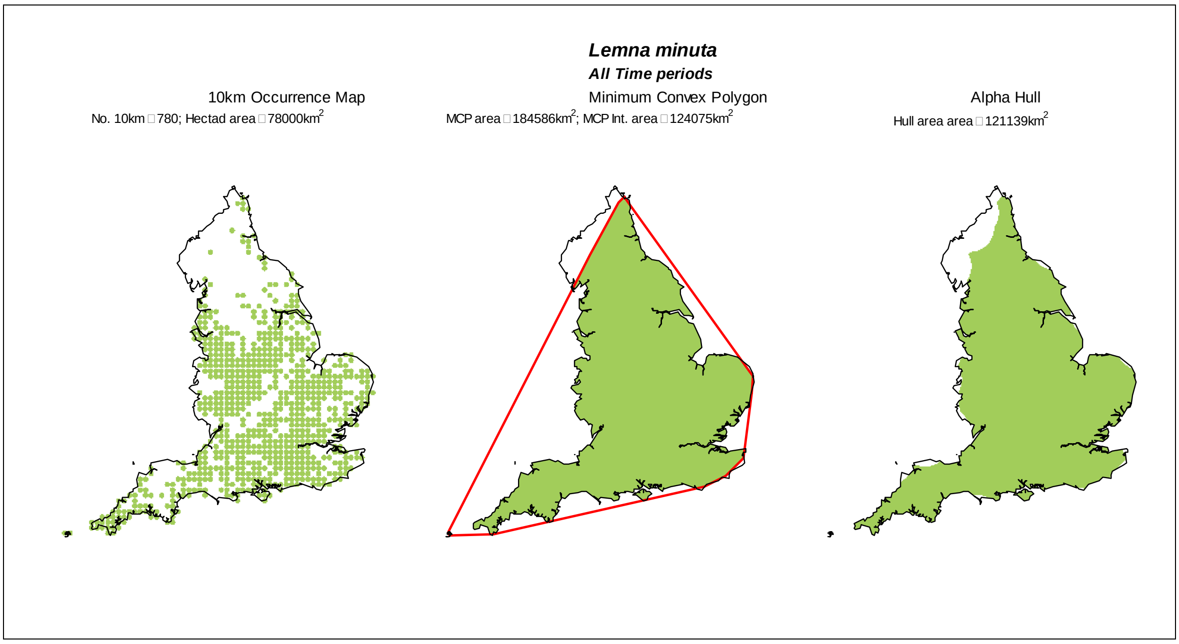


Figure B.46 Area of extent maps for *Lemna minuta* using all records

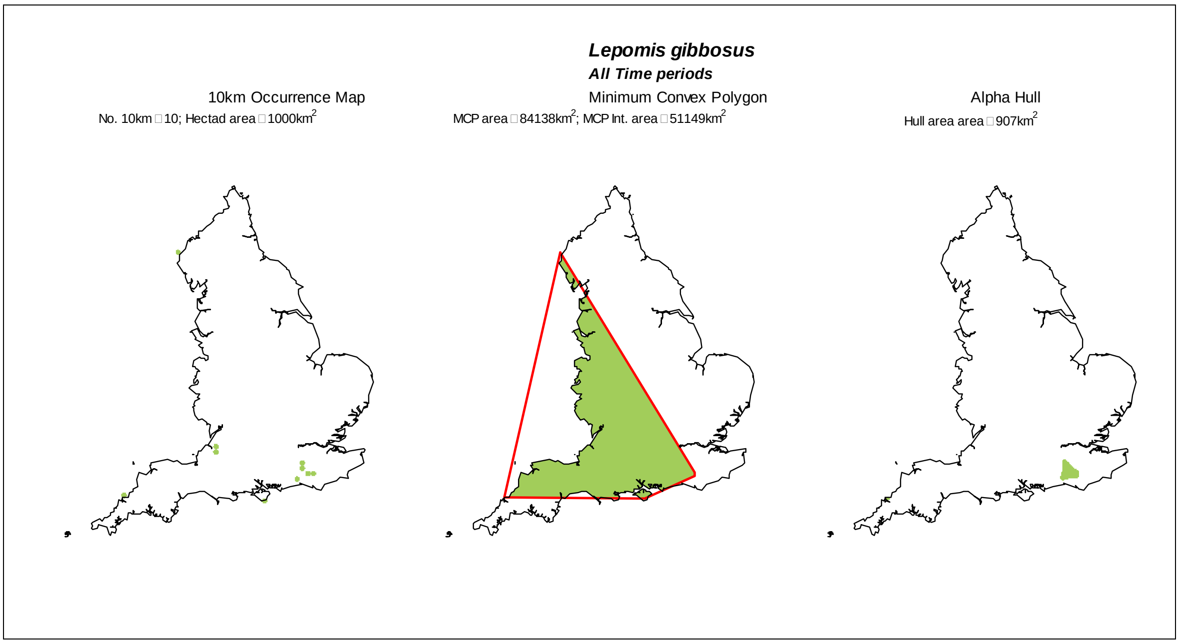


Figure B.47 Area of extent maps for *Lepomis gibbosus* using all records

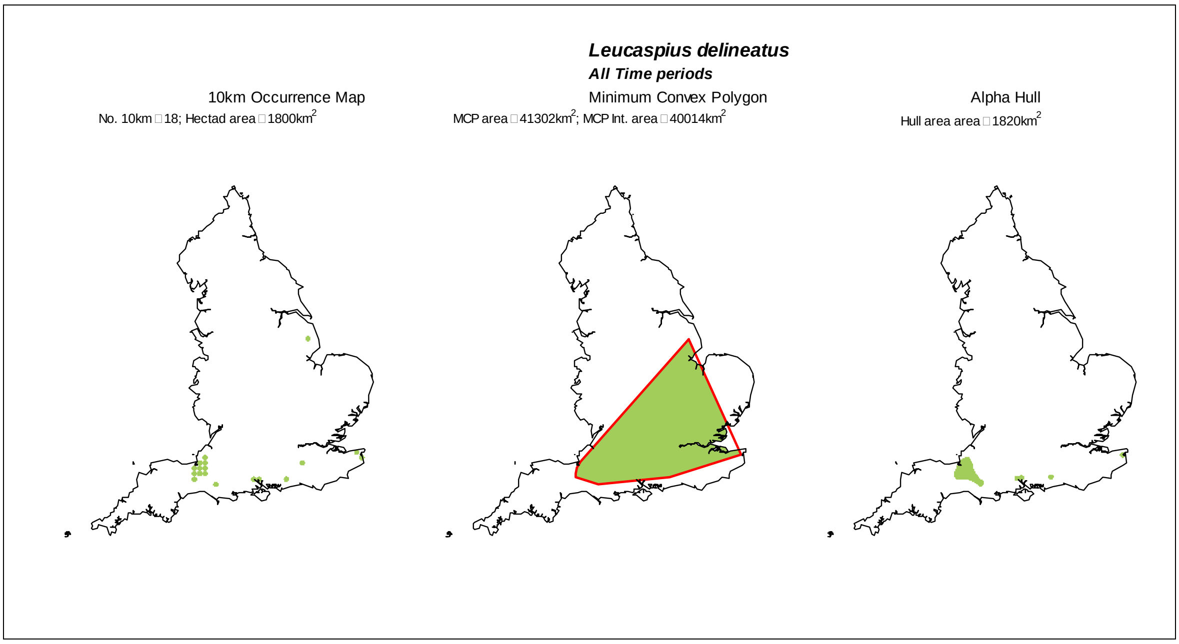


Figure B.48 Area of extent maps for *Leucaspius delineatus* using all records

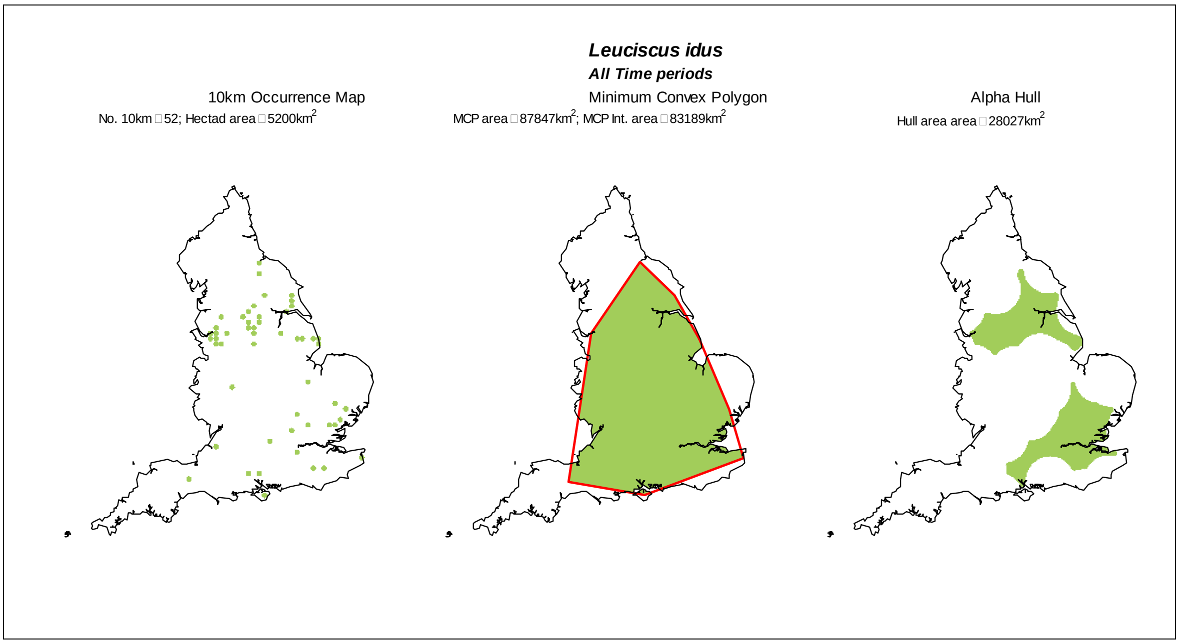


Figure B.49 Area of extent maps for *Leuciscus idus* using all records

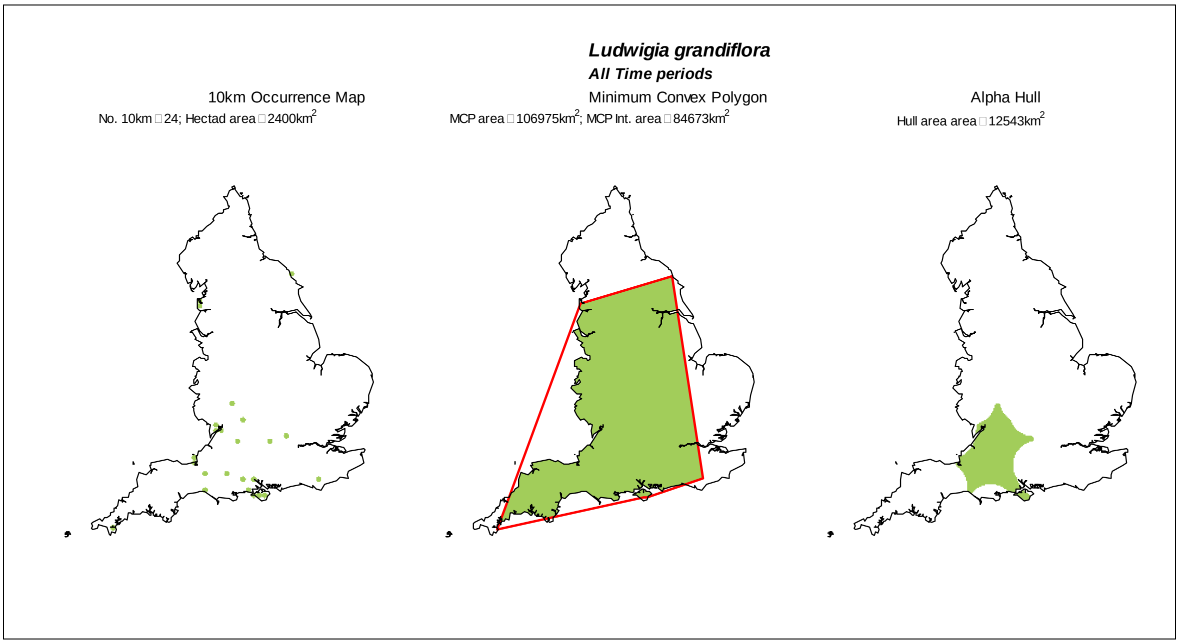


Figure B.50 Area of extent maps for *Ludwigia grandiflora* using all records

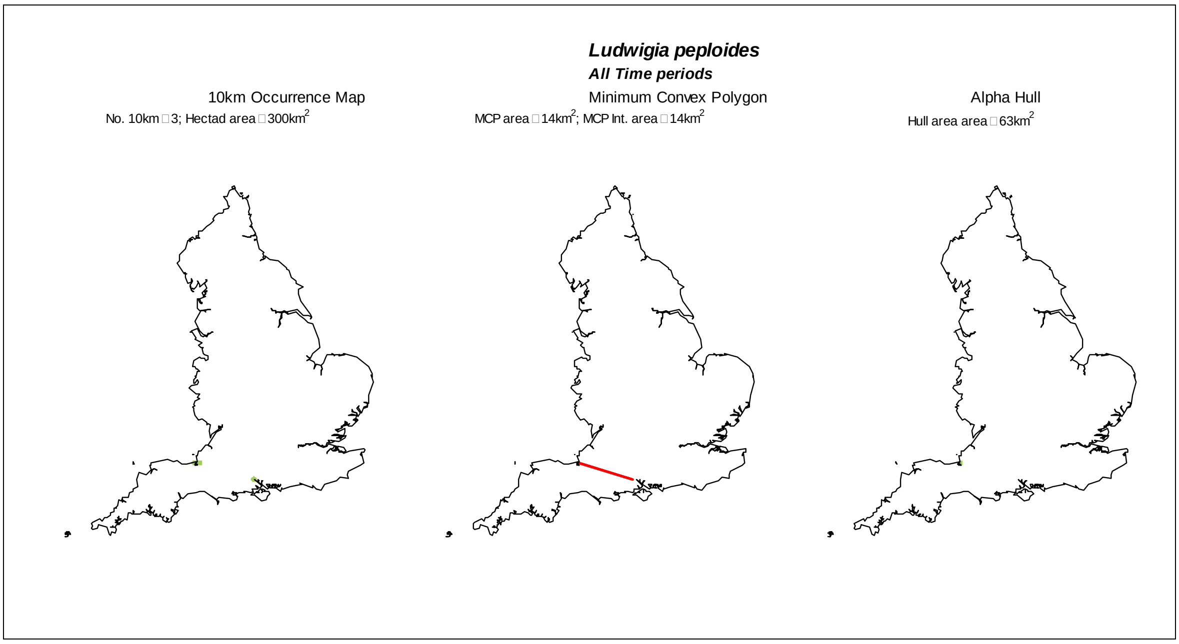


Figure B.51 Area of extent maps for *Ludwigia peploides* using all records

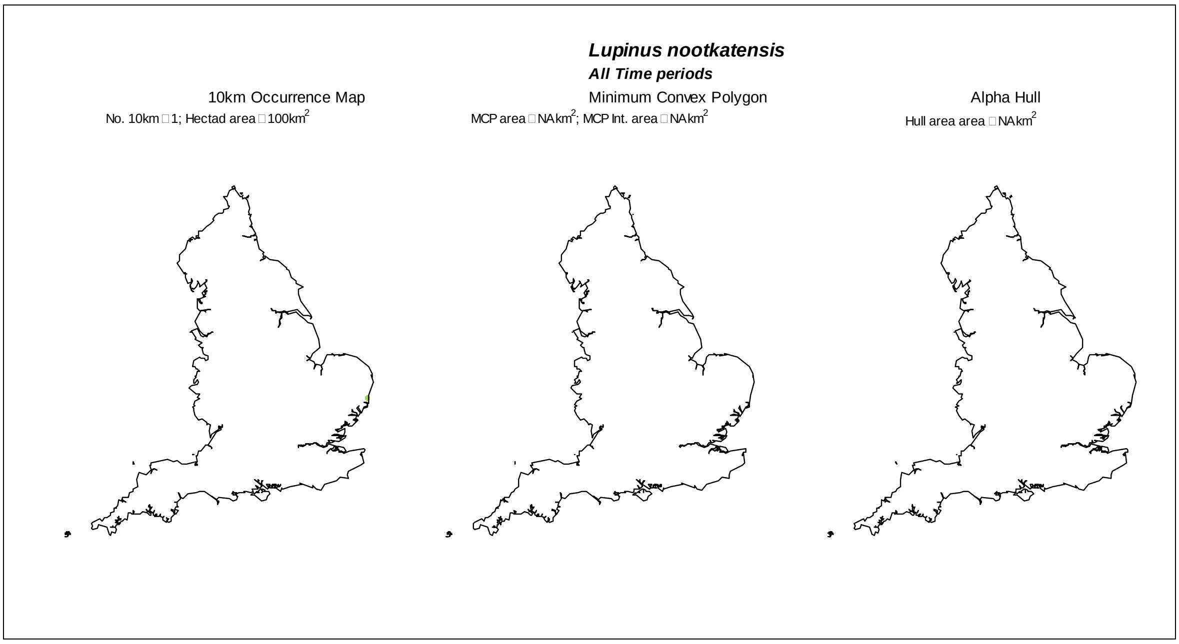


Figure B.52 Area of extent maps for *Lupinus nootkatensis* using all records

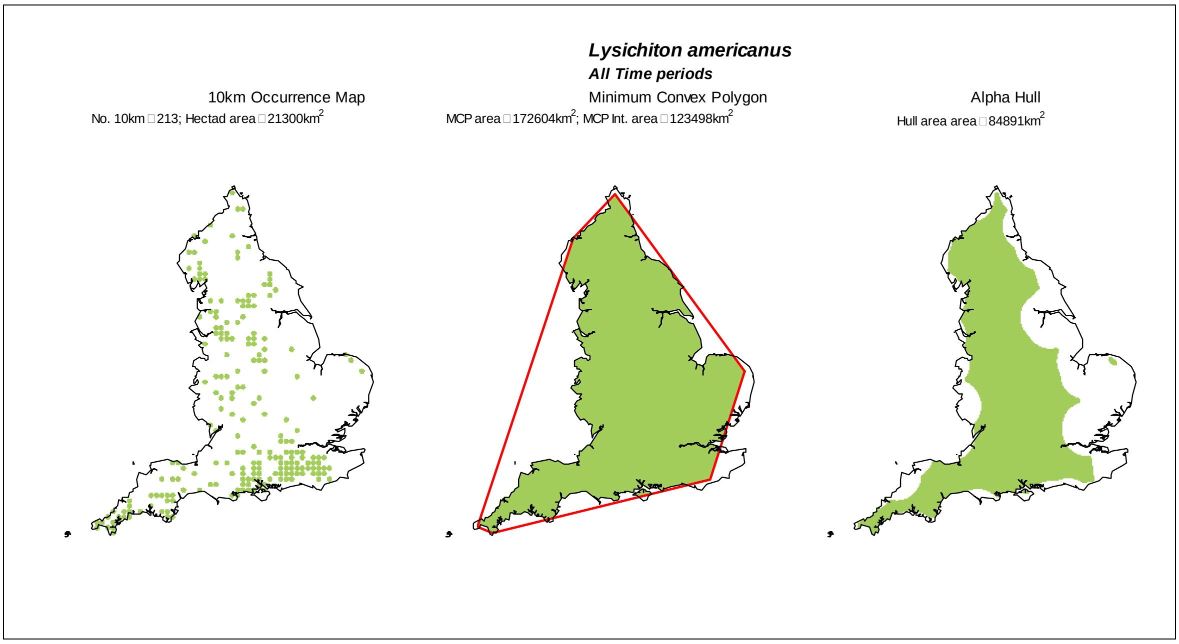


Figure B.53 Area of extent maps for *Lysichiton americanus* using all records

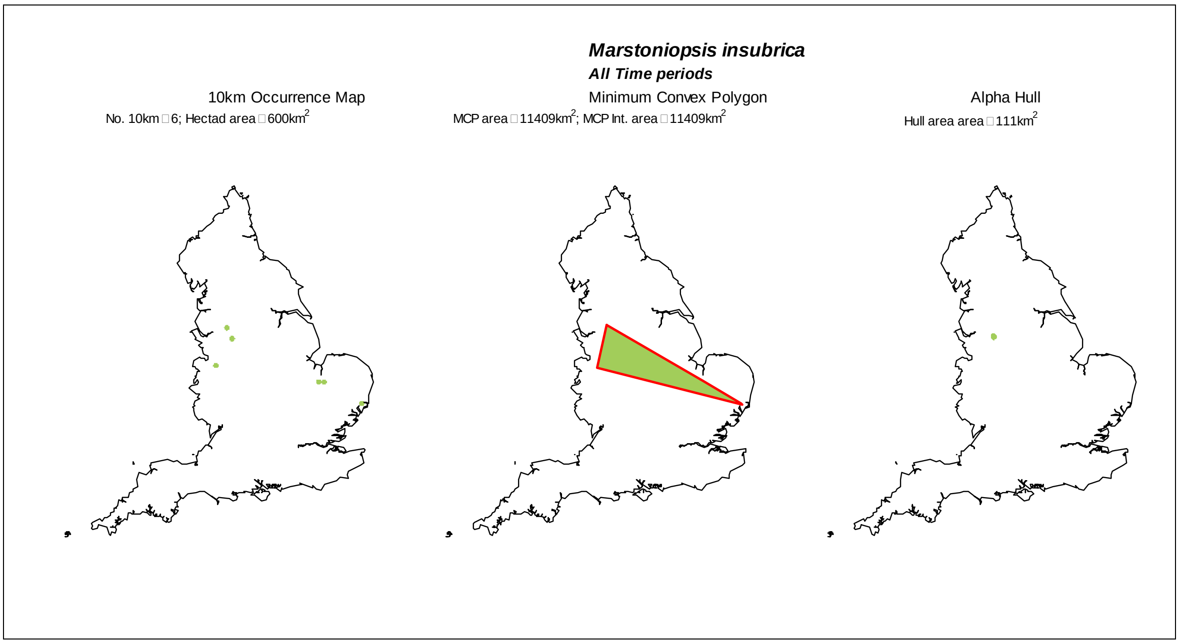


Figure B.54 Area of extent maps for *Marstoniopsis insubrica* using all records

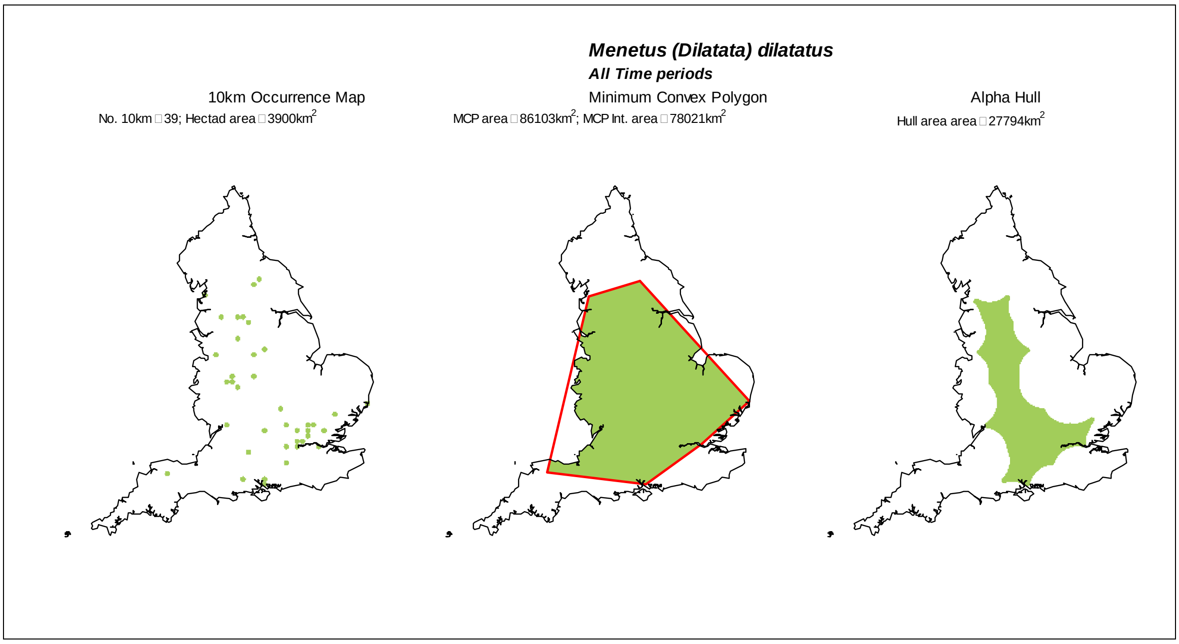


Figure B.55 Area of extent maps for *Menetus (Dilatata) dilatatus* using all records

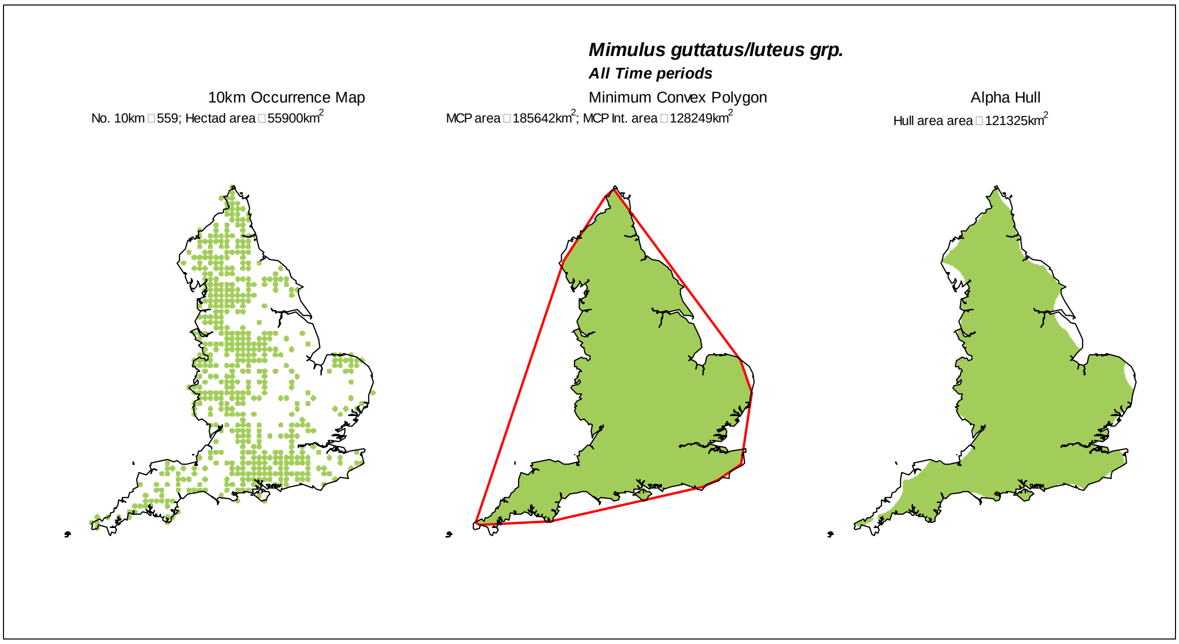


Figure B.56 Area of extent maps for *Mimulus guttatus/luteus* group using all records

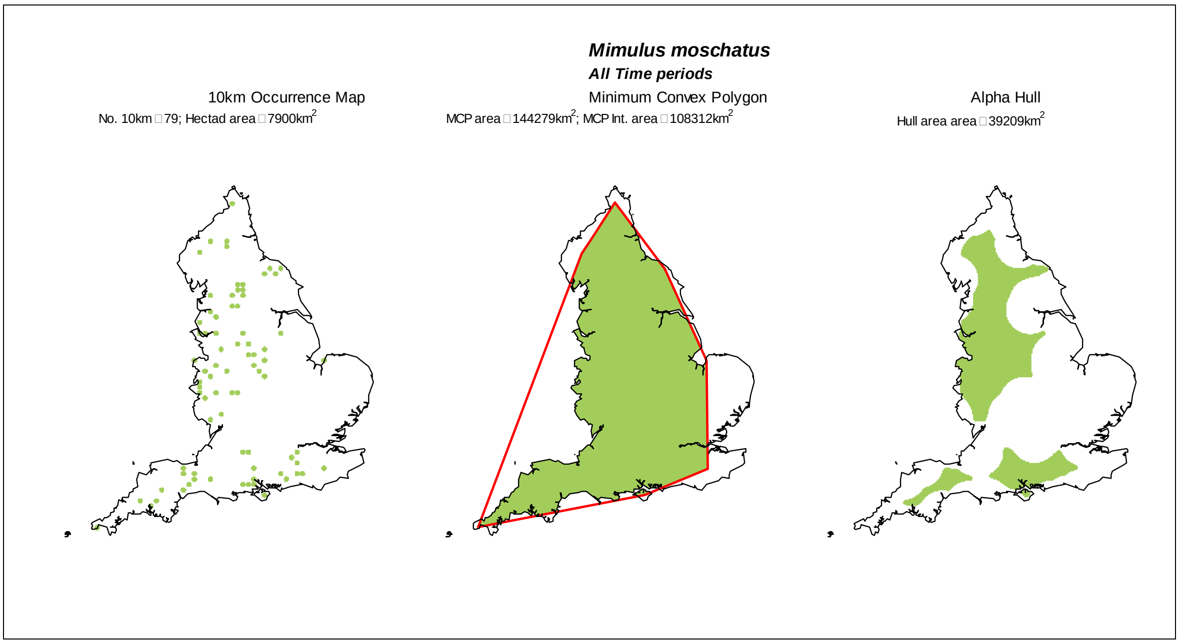


Figure B.57 Area of extent maps for *Mimulus moschatus* using all records

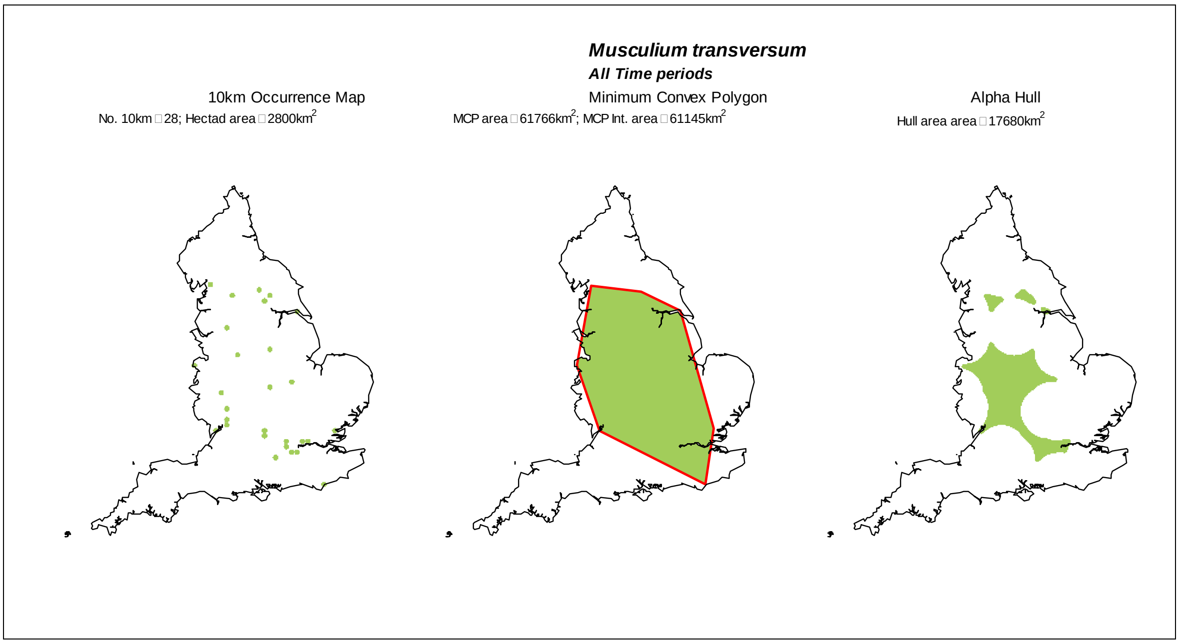


Figure B.58 Area of extent maps for *Musculium transversum* using all records

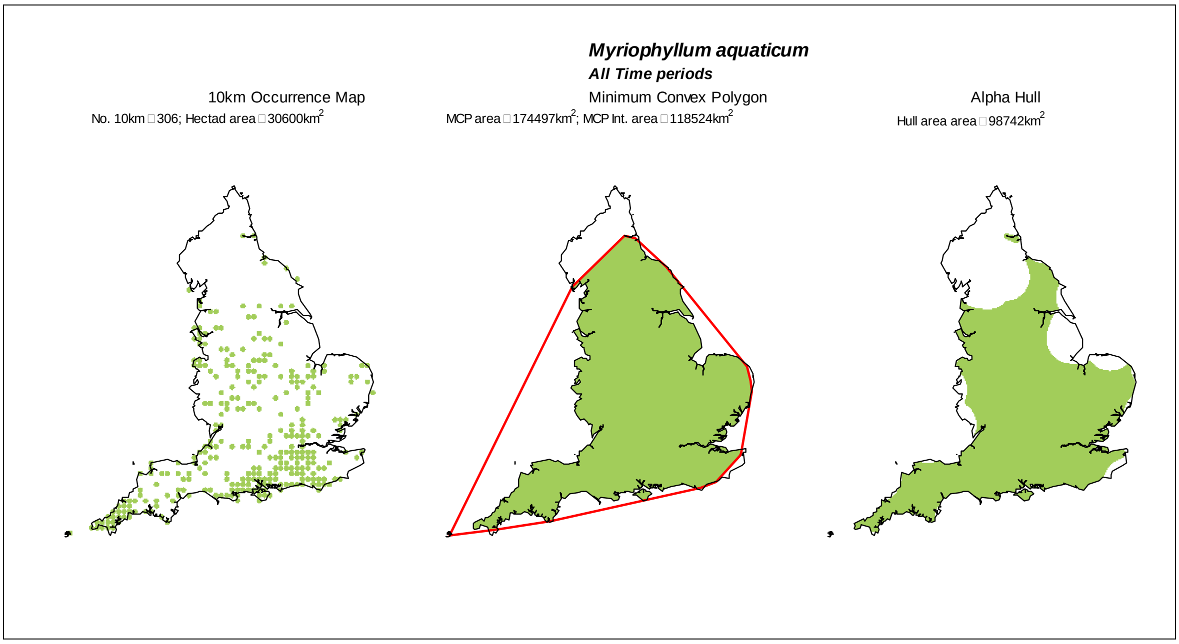


Figure B.59 Area of extent maps for *Myriophyllum aquaticum* using all records

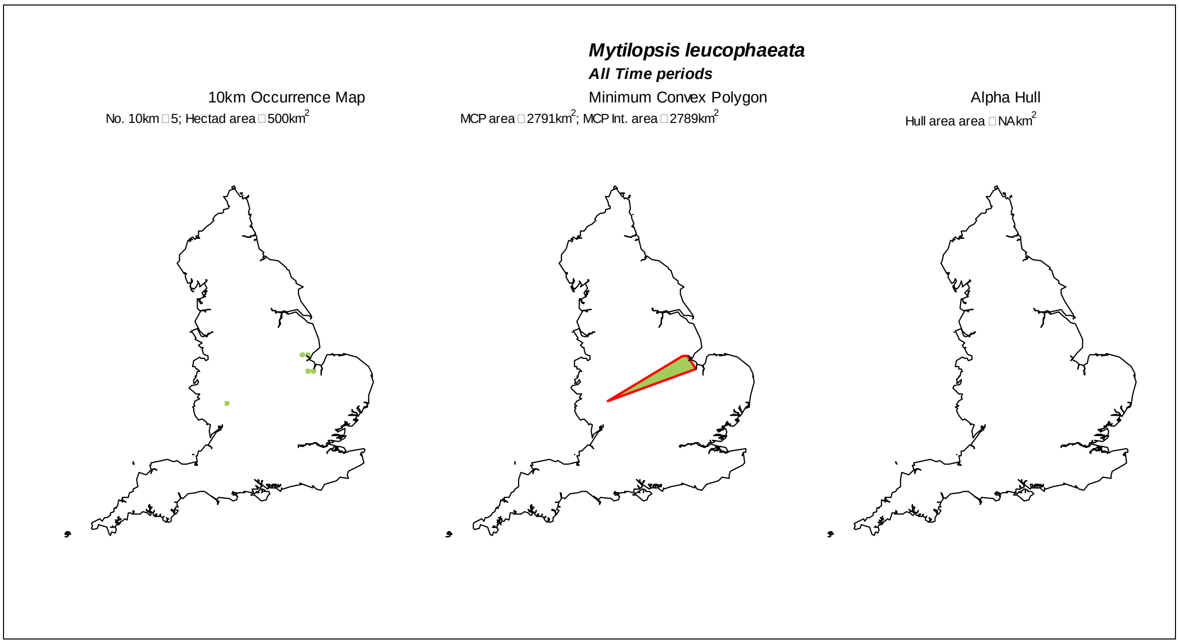


Figure B.60 Area of extent maps for *Mytilopsis leucophaeata* using all records

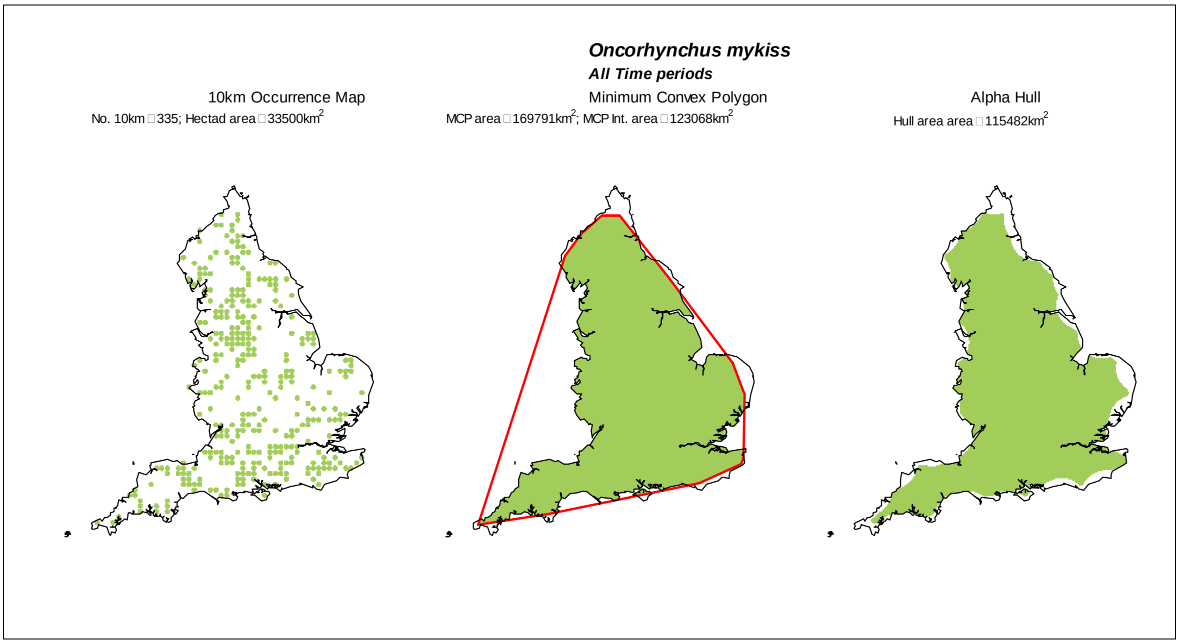


Figure B.61 Area of extent maps for *Oncorhynchus mykiss* using all records

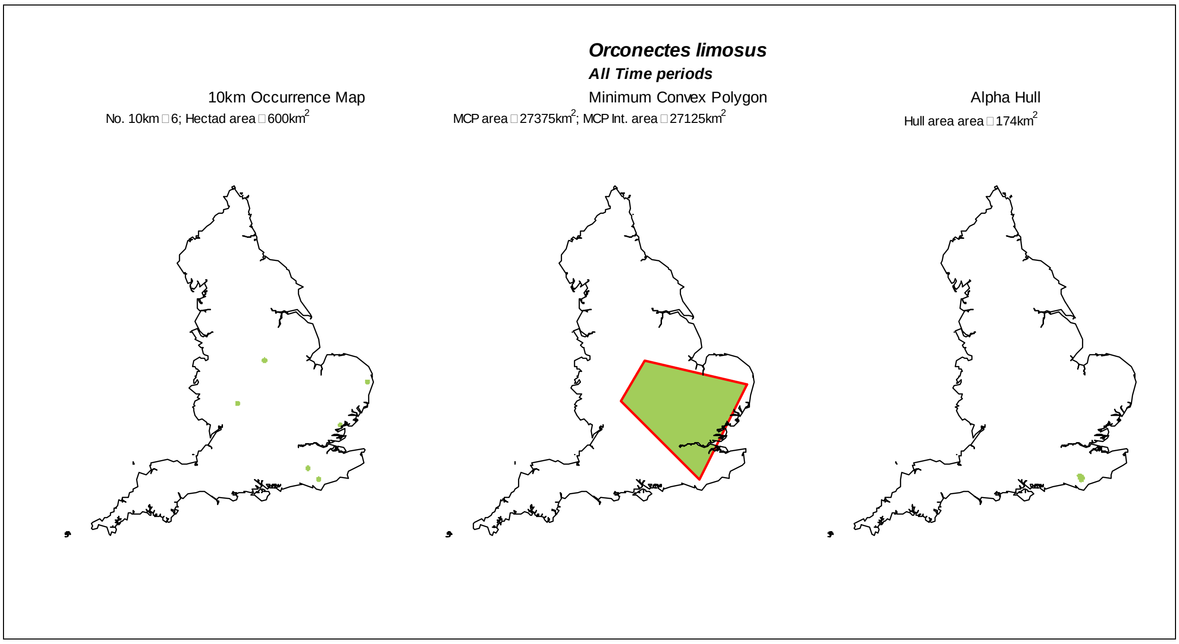


Figure B.62 Area of extent maps for *Orconectes limosus* using all records

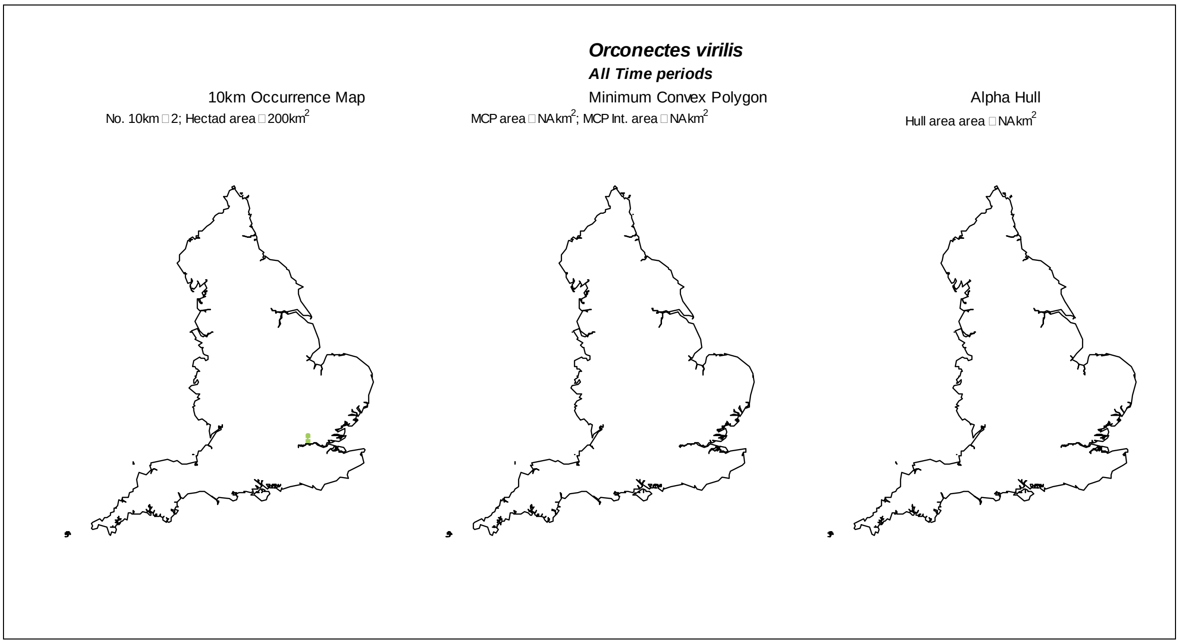


Figure B.63 Area of extent maps for *Orconectes virilis* using all records

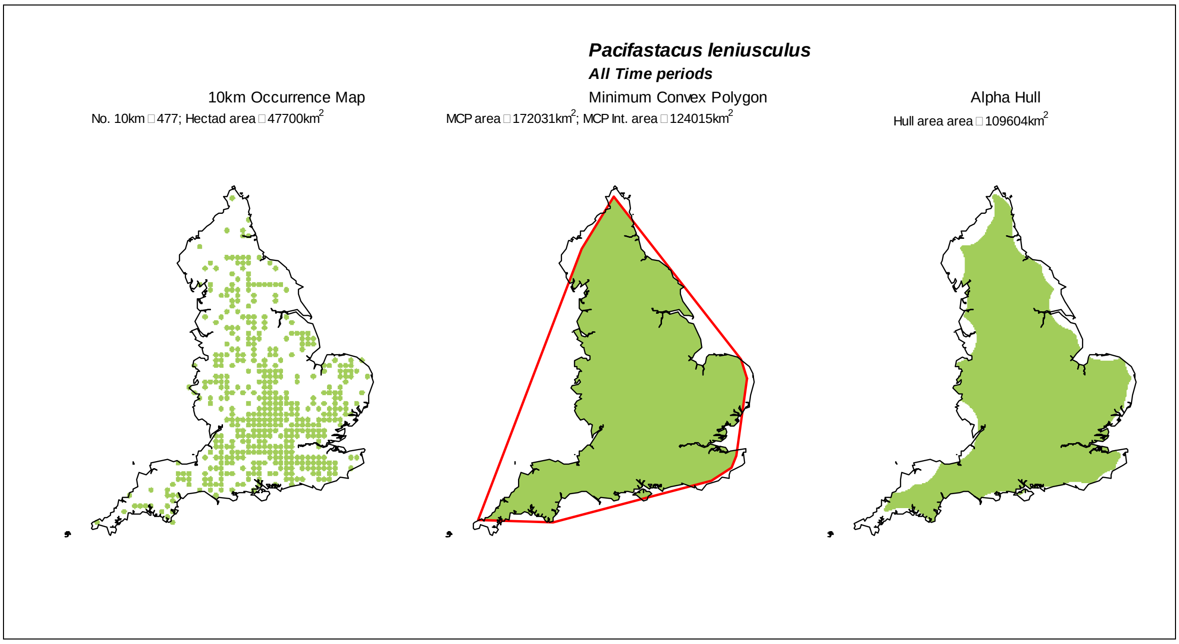


Figure B.64 Area of extent maps for *Pacifastacus leniusculus* using all records

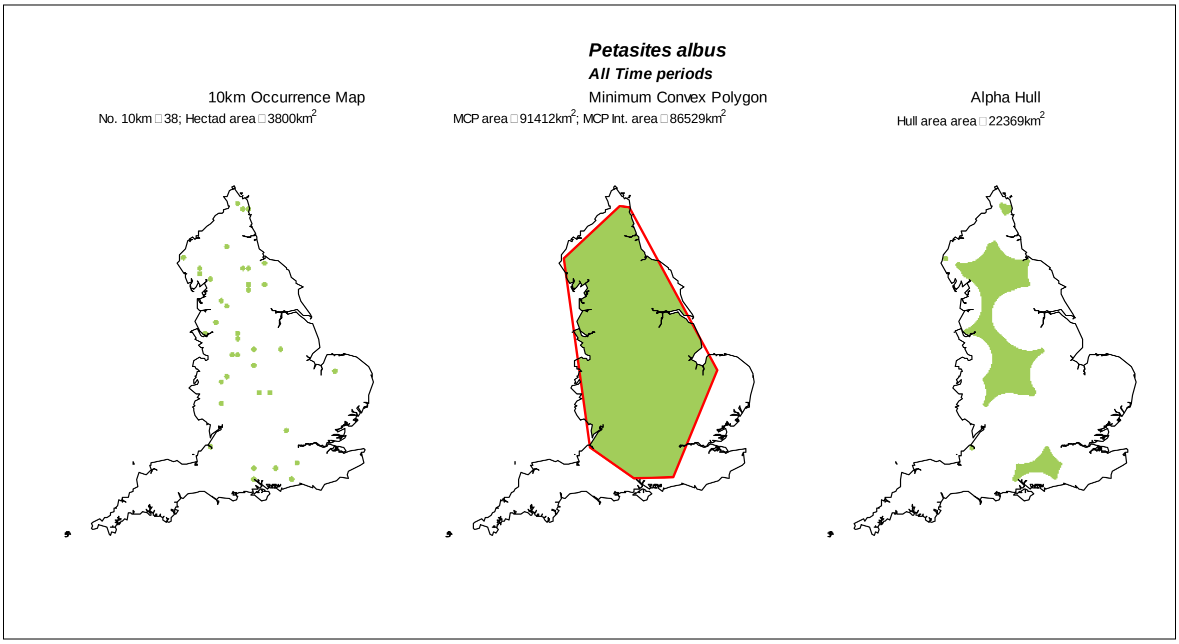


Figure B.65 Area of extent maps for *Petasites albus* using all records

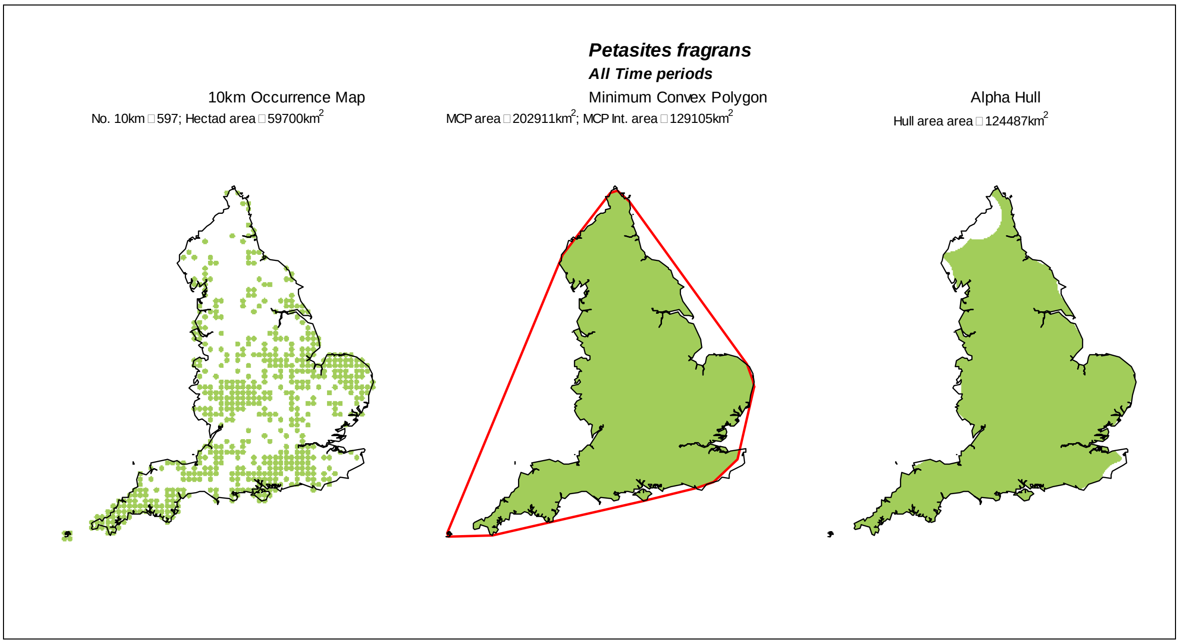


Figure B.66 Area of extent maps for *Petasites fragrans* using all records

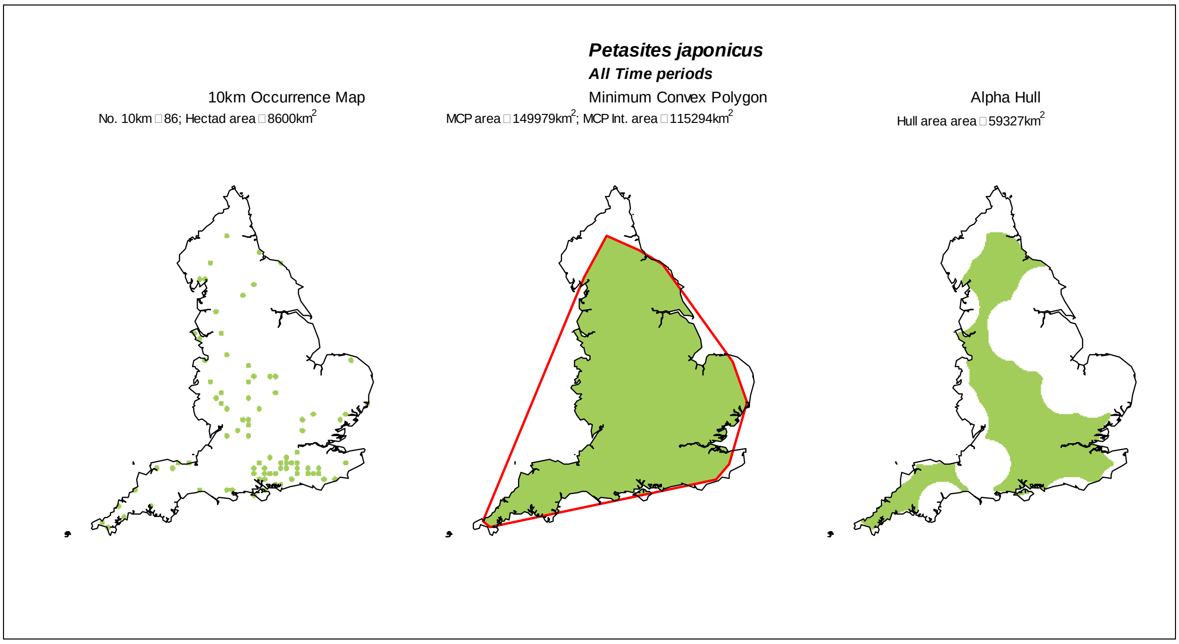


Figure B.67 Area of extent maps for *Petasites japonicus* using all records

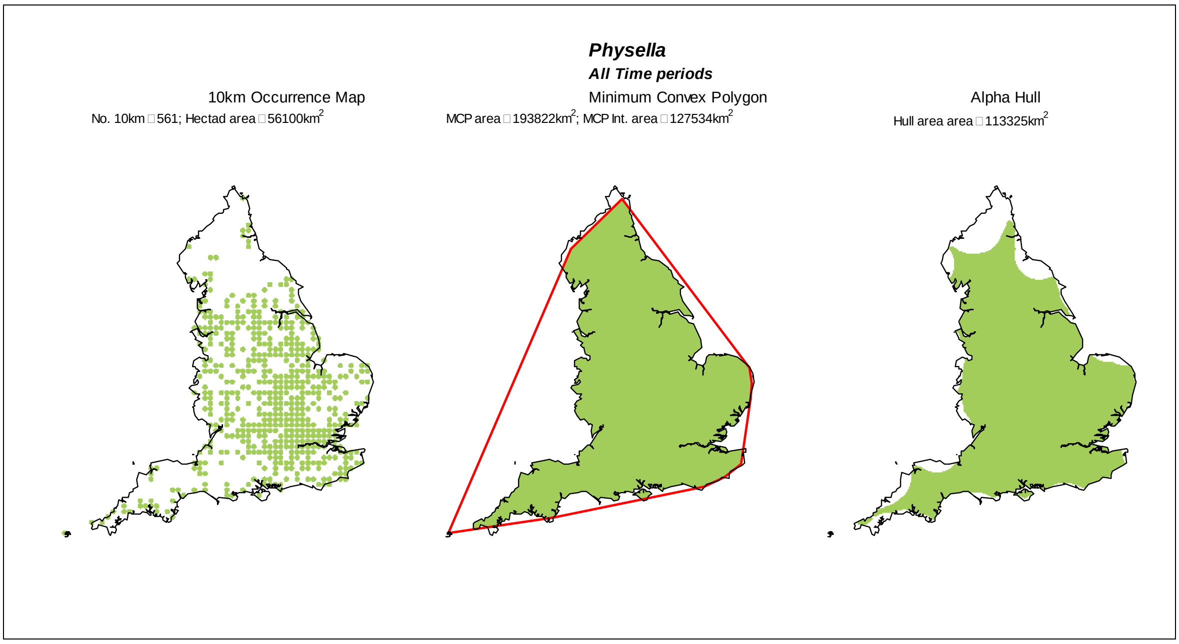


Figure B.68 Area of extent maps for *Physella* using all records

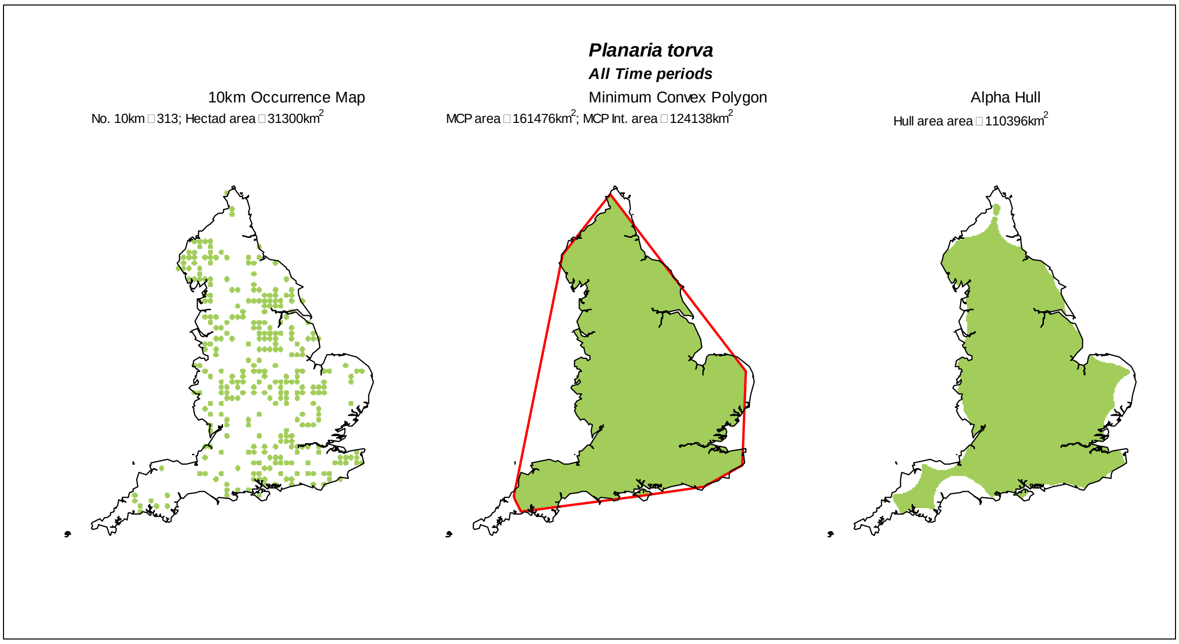


Figure B.69 Area of extent maps for *Planaria torva* using all records

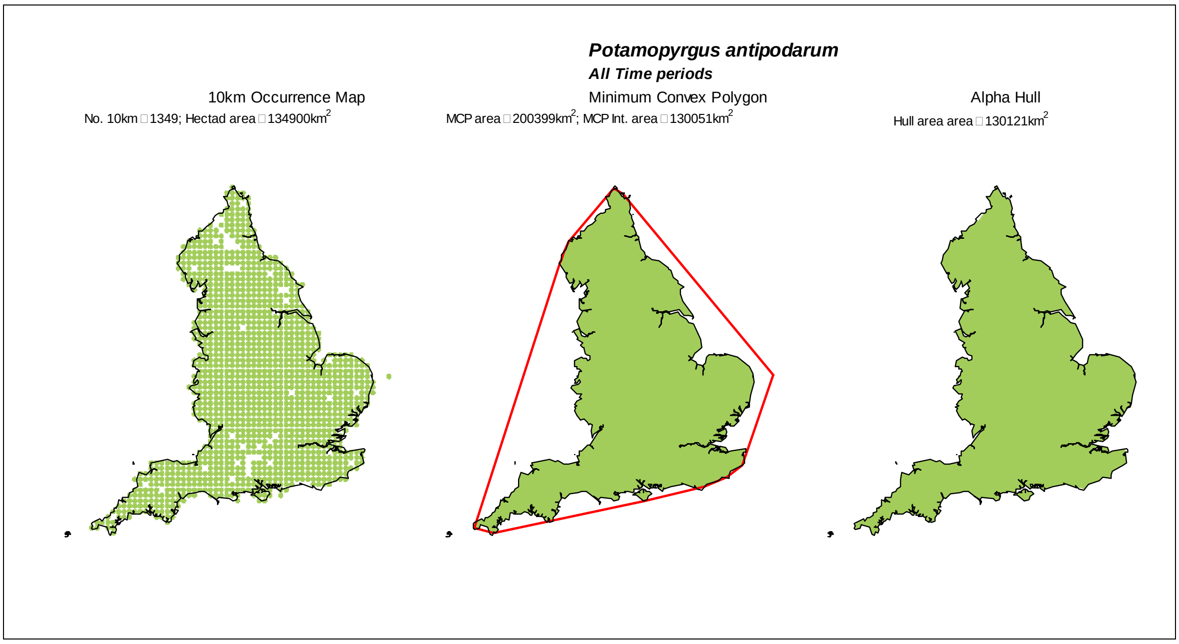


Figure B.70 Area of extent maps for *Potamopyrgus antipodarum* using all records

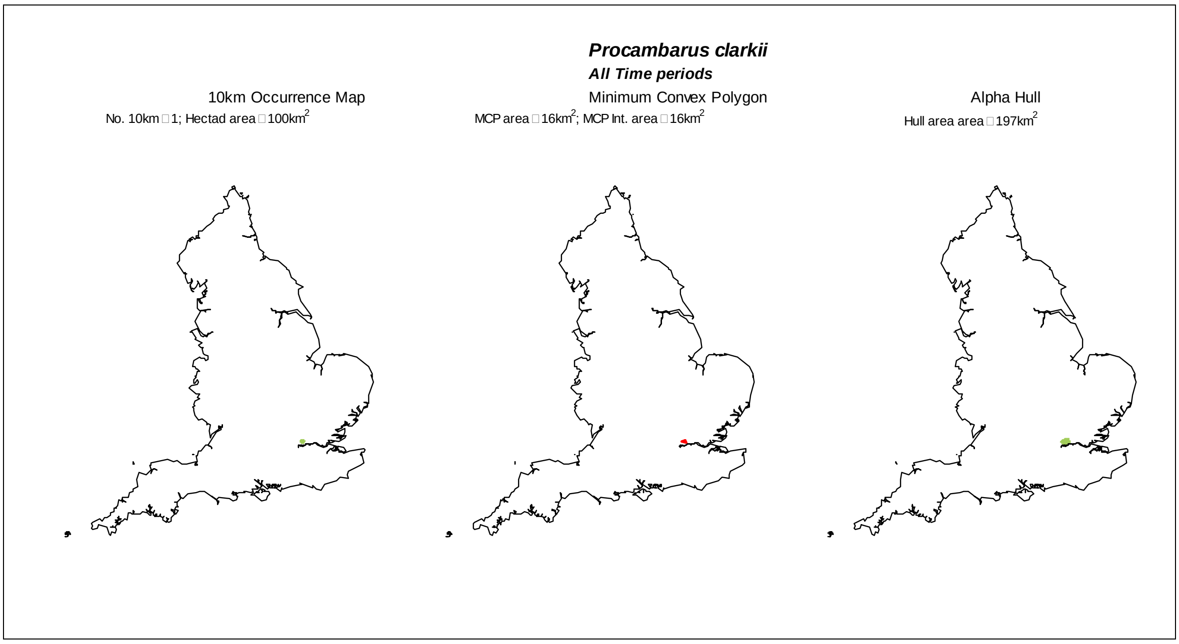


Figure B.71 Area of extent maps for *Procambarus clarkii* using all records

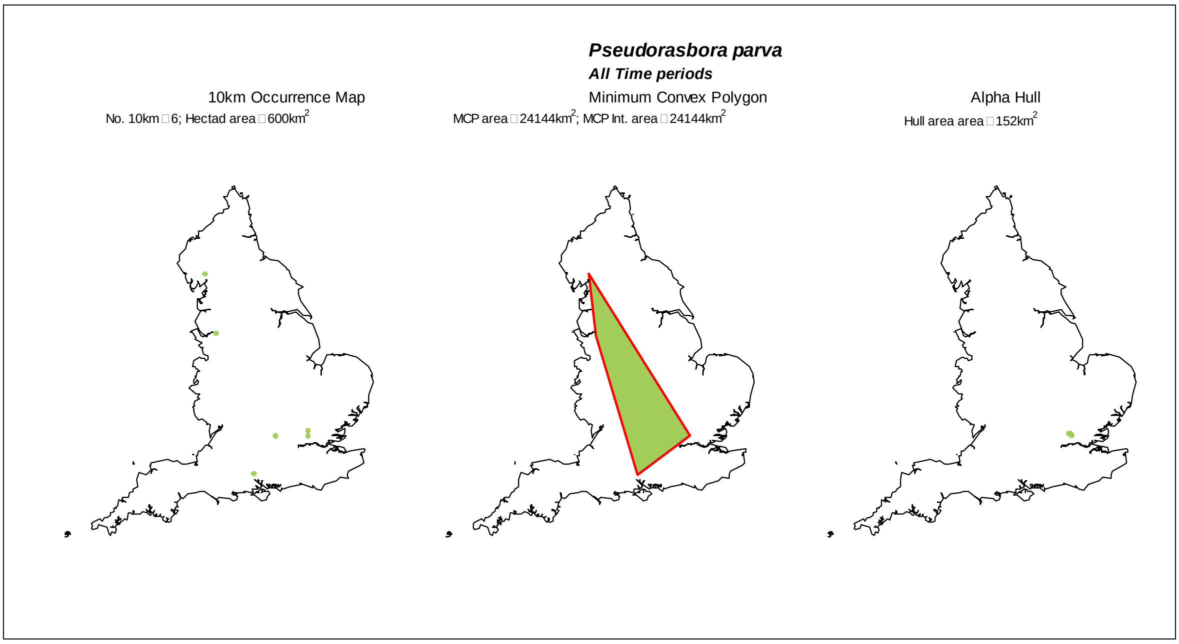


Figure B.72 Area of extent maps for *Pseudorasbora parva* using all records

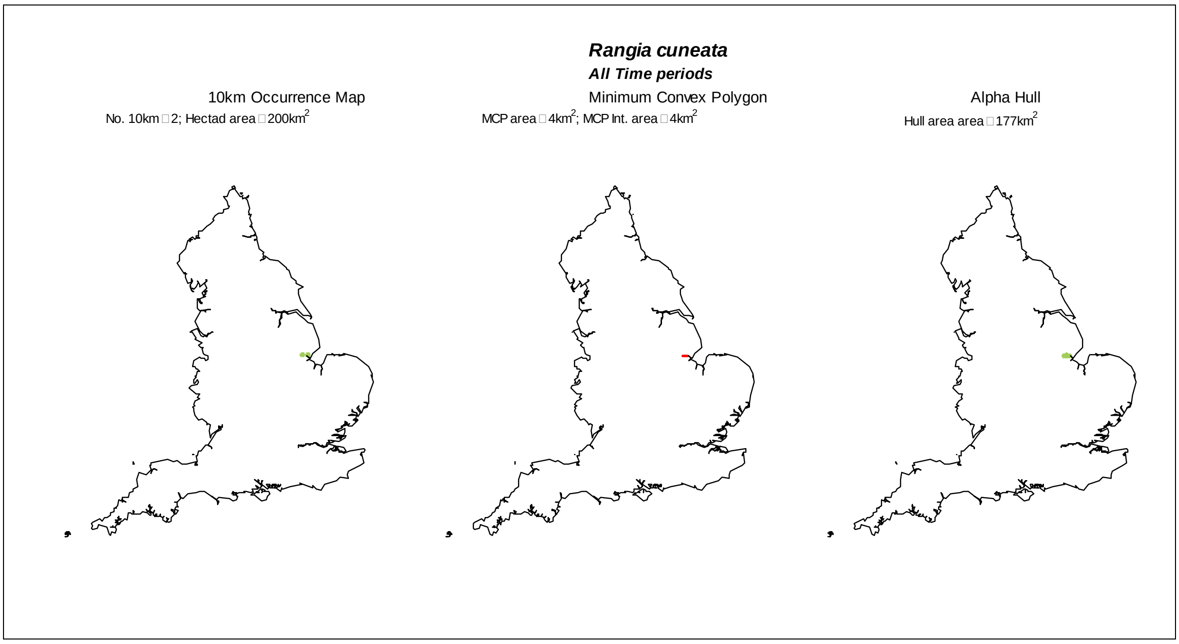


Figure B.73 Area of extent maps for *Rangia cuneata* using all records

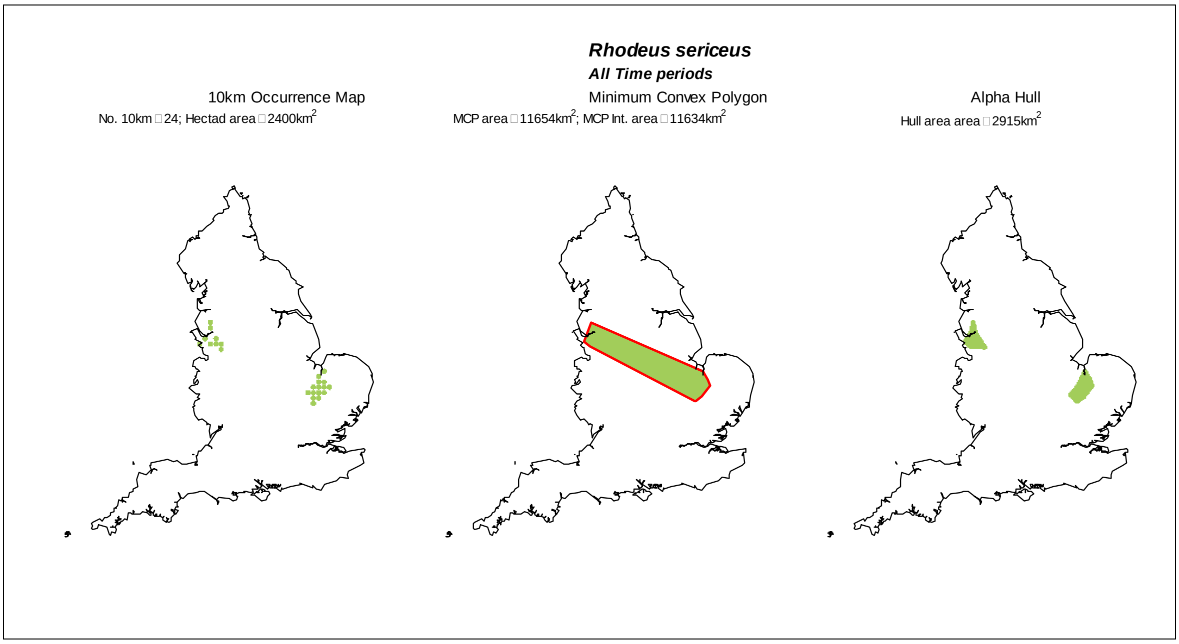


Figure B.74 Area of extent maps for *Rhodeus sericeus* using all records

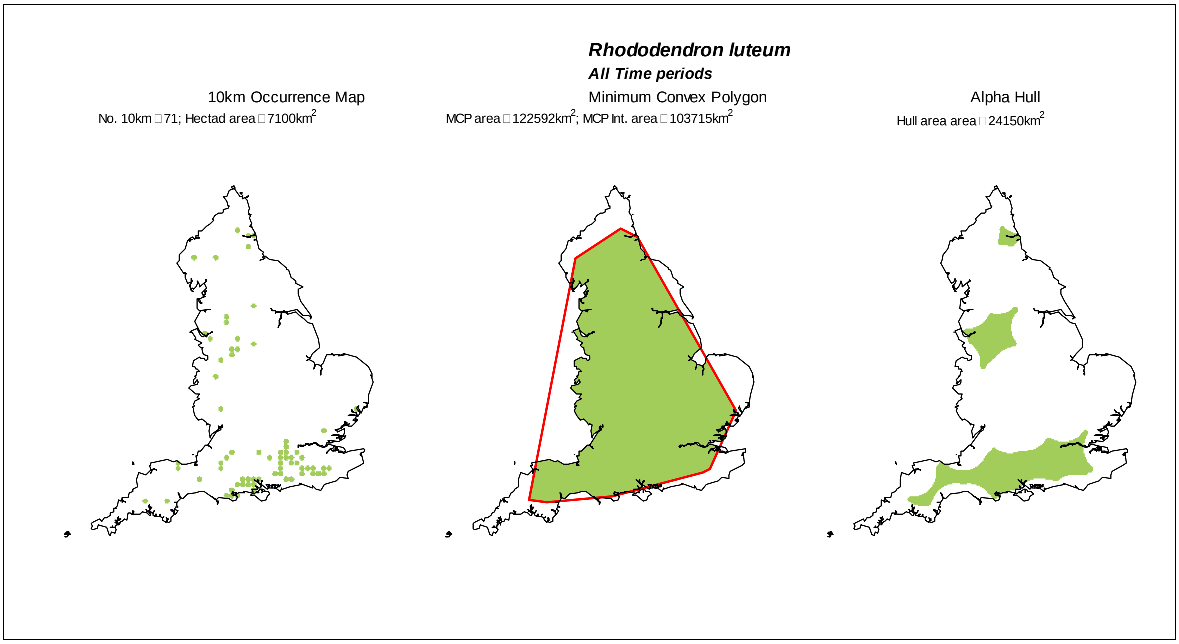


Figure B.75 Area of extent maps for *Rhododendron luteum* using all records

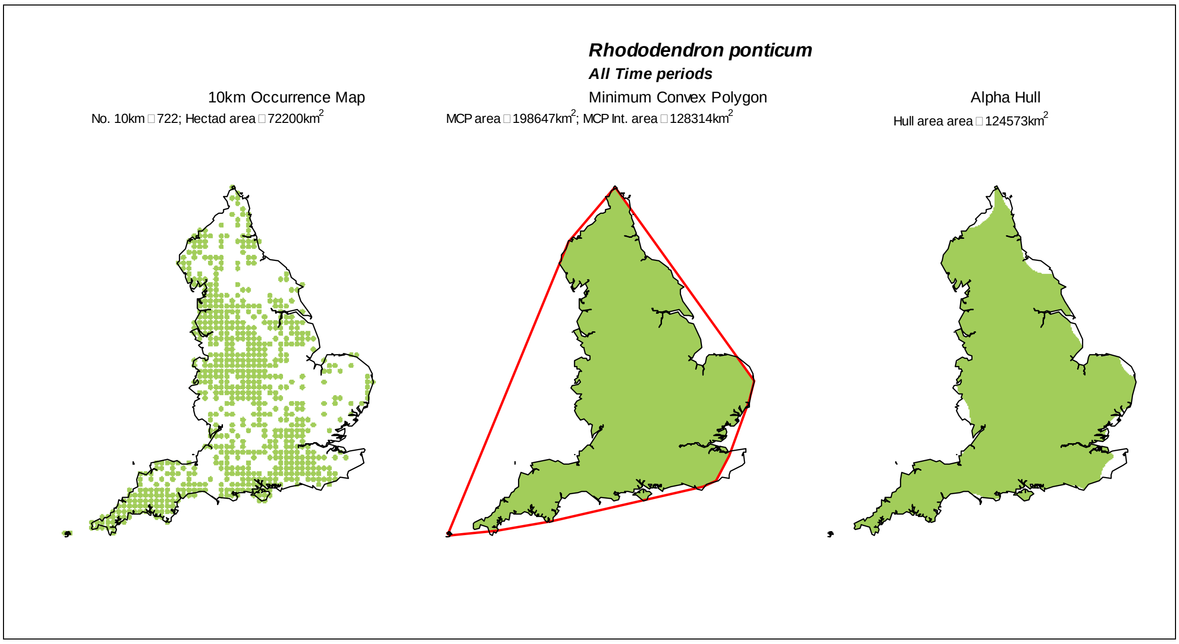


Figure B.76 Area of extent maps for *Rhododendron ponticum* using all records

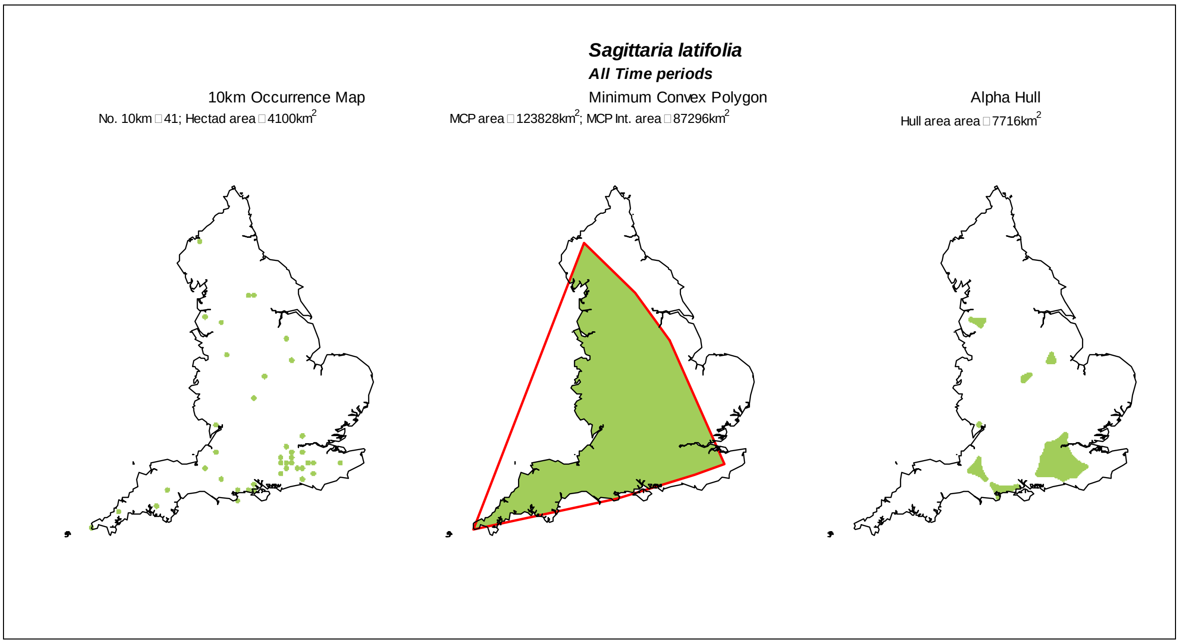


Figure B.77 Area of extent maps for *Sagittaria latifolia* using all records

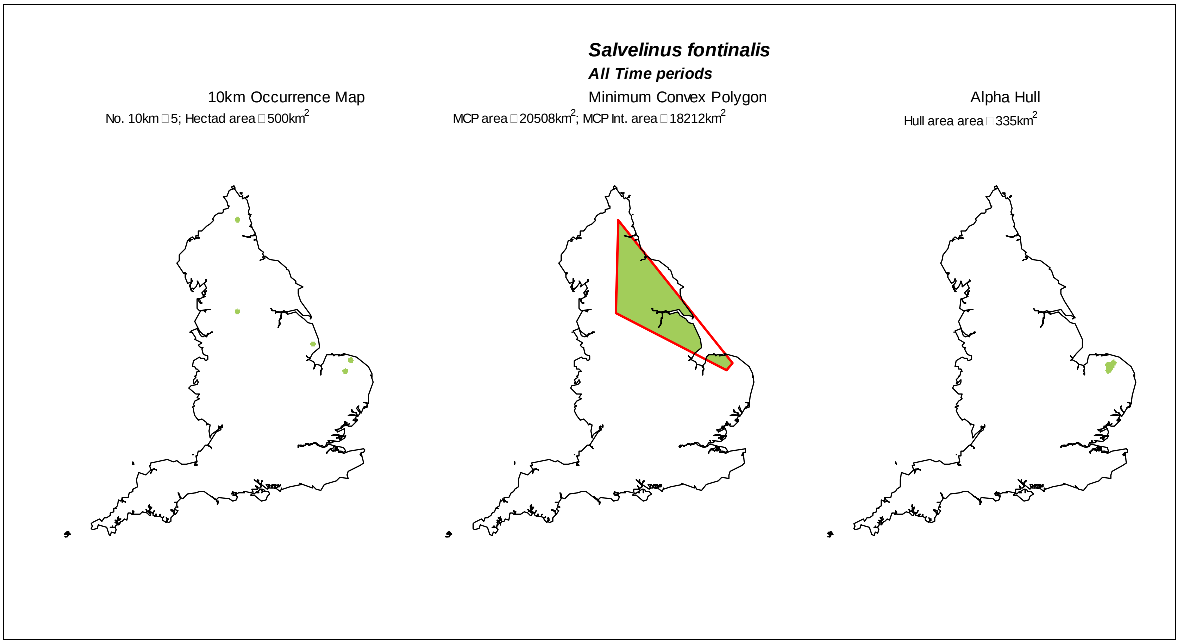


Figure B.78 Area of extent maps for *Salvelinus fontinalis* using all records

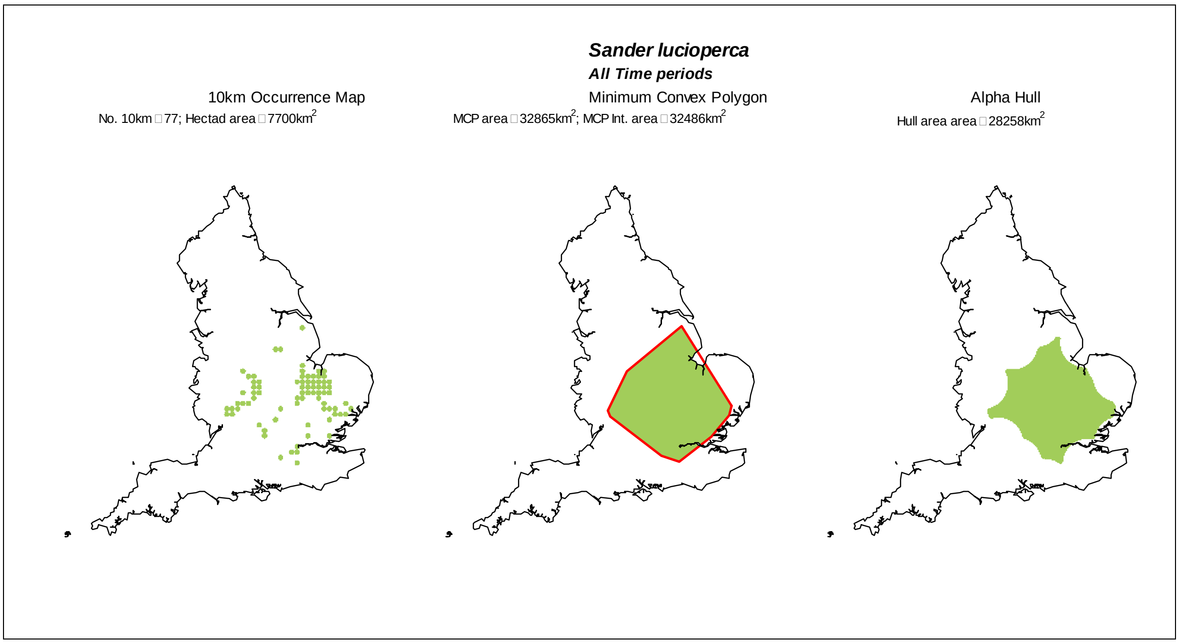


Figure B.79 Area of extent maps for *Sander lucioperca* using all records

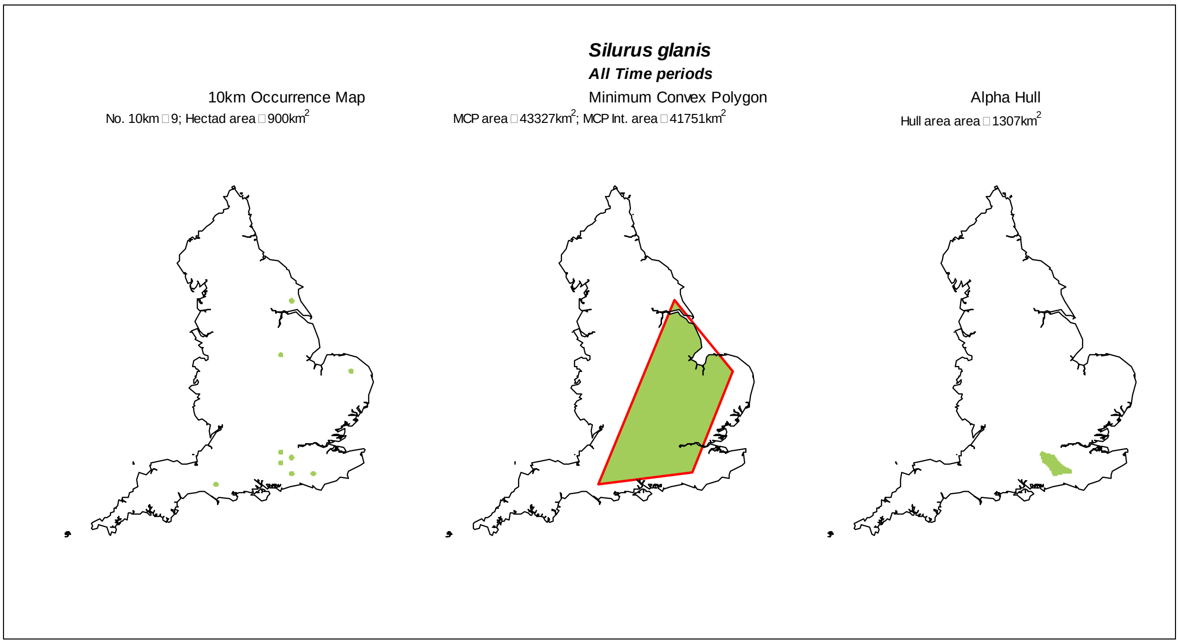


Figure B.80 Area of extent maps for *Silurus glanis* using all records

Appendix C: Decadal change in area of extent

For each species, the area of extent is shown by decade. For each decade, the first map shows the 10km occurrence data, the second map shows the MCP (outlined by a red line) and its intersection with the land (green filled region), and the third map shows the alpha hull and its intersection with the land (green filled region).

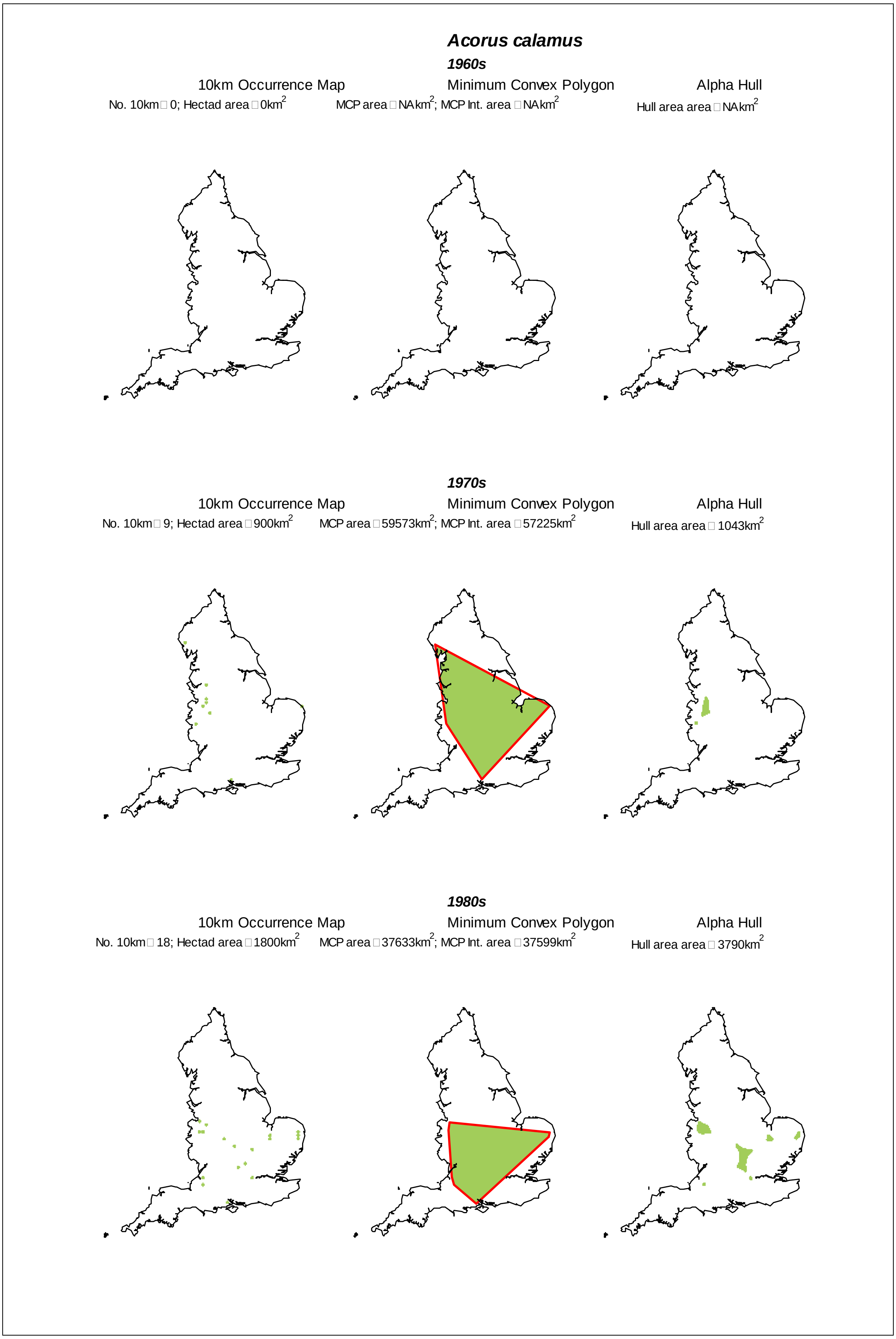


Figure C.1a Change in area of extent for *Acorus calamus* by decade (1960s to 1980s)

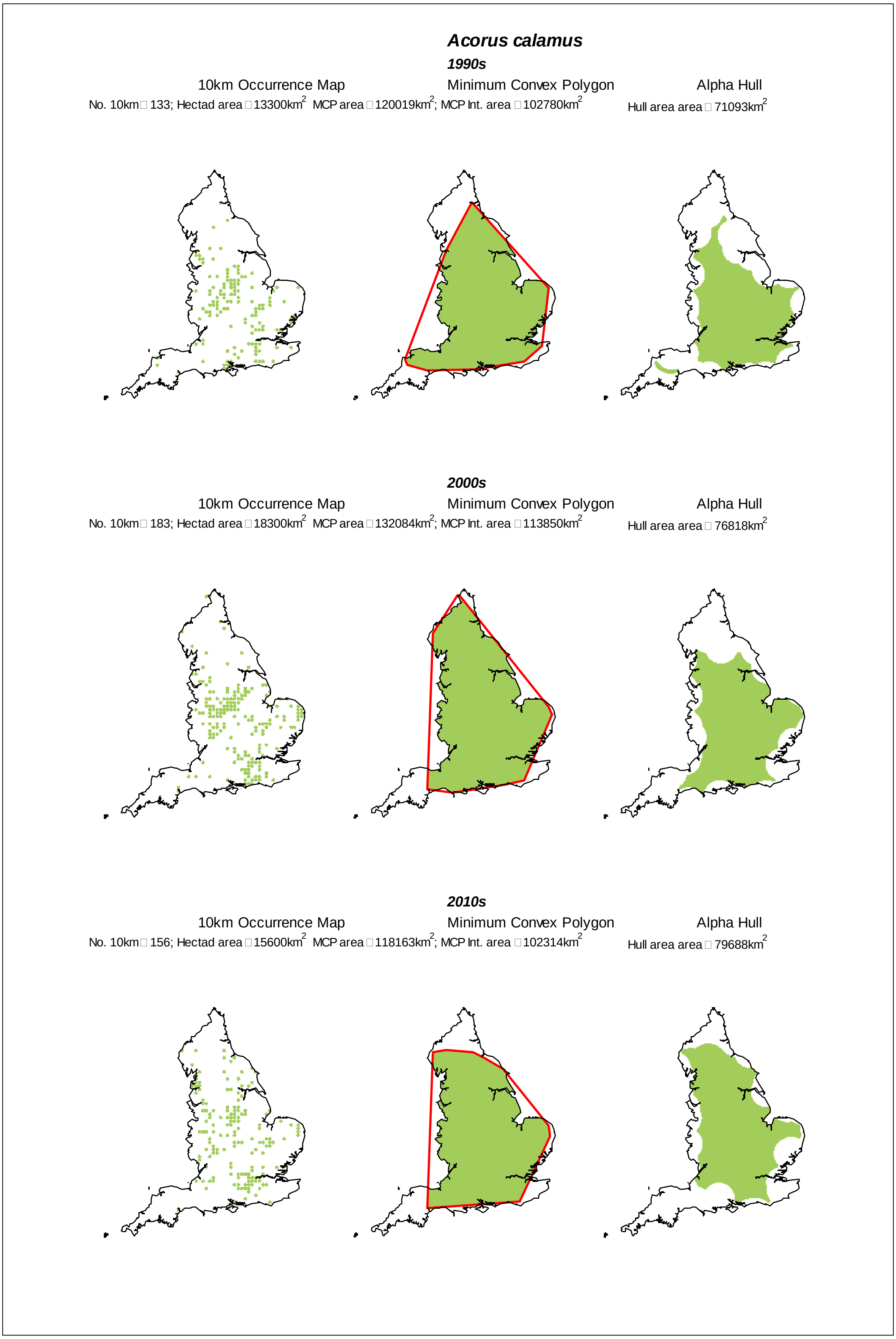


Figure C.1b Change in area of extent for *Acorus calamus* by decade (1990s to 2010s)

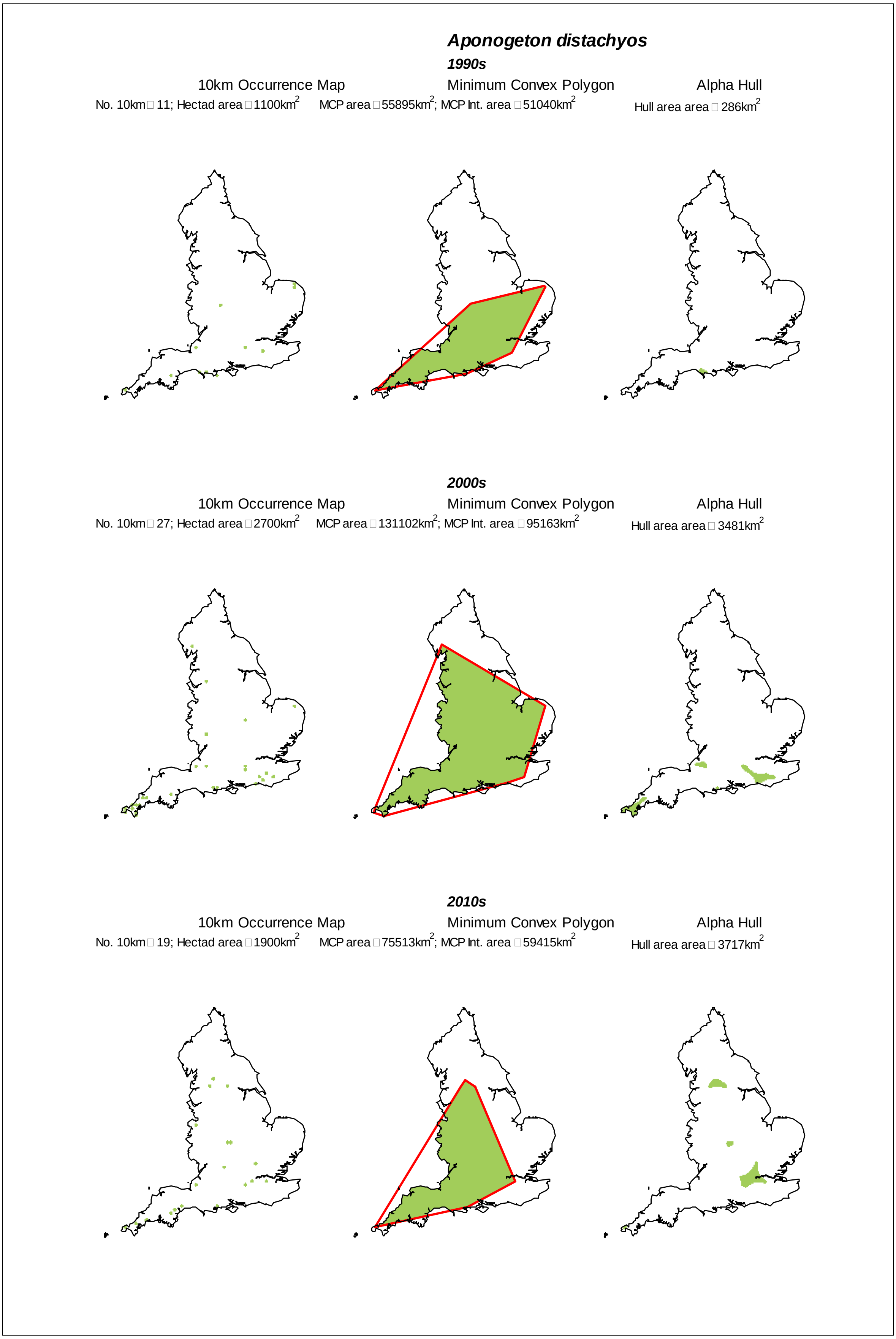


Figure C.2 Change in area of extent for *Aponogeton distachyos* by decade ((1990s to 2010s)

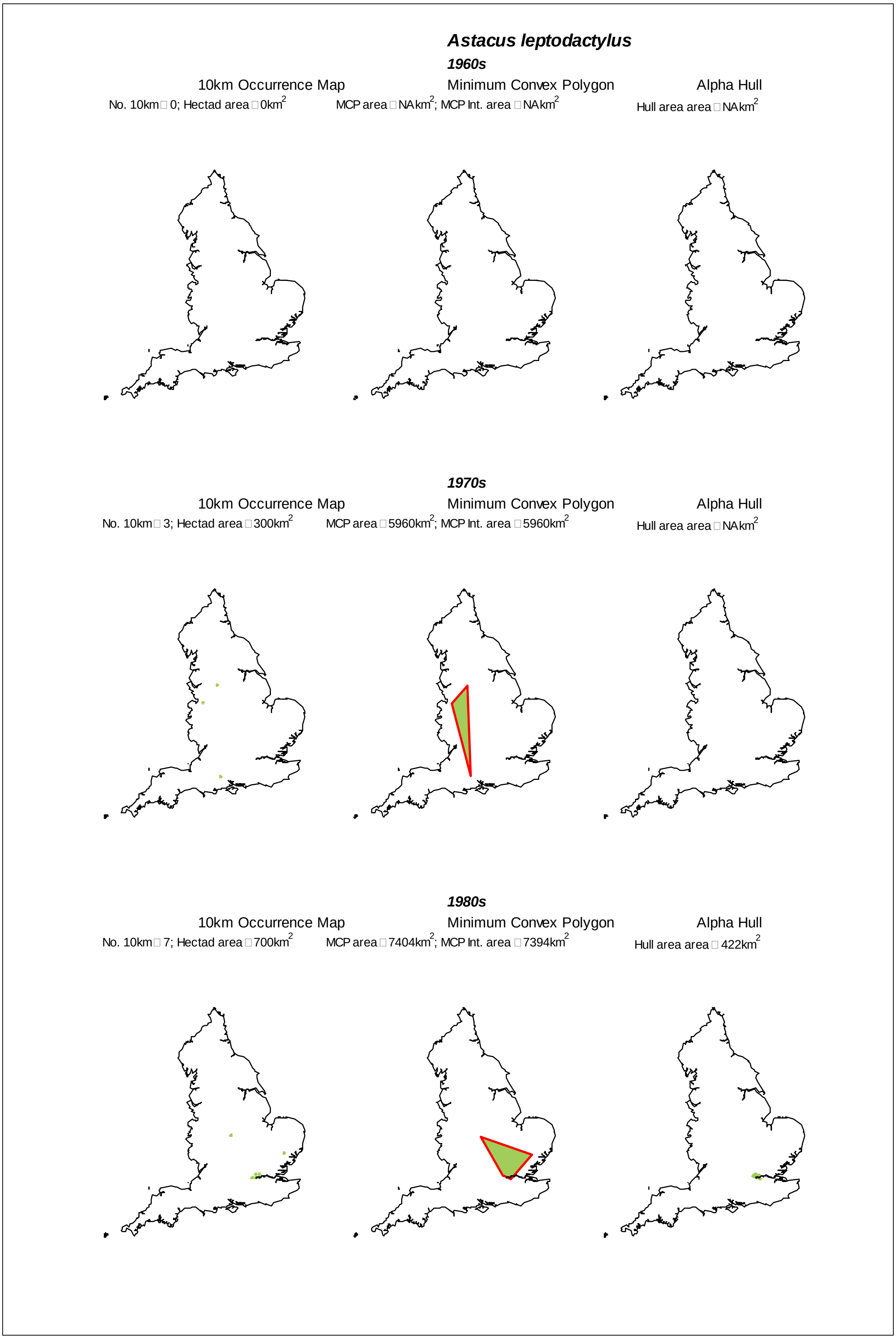


Figure C.3a Change in area of extent for *Astacus leptodactylus* by decade (1960s to 1980s)

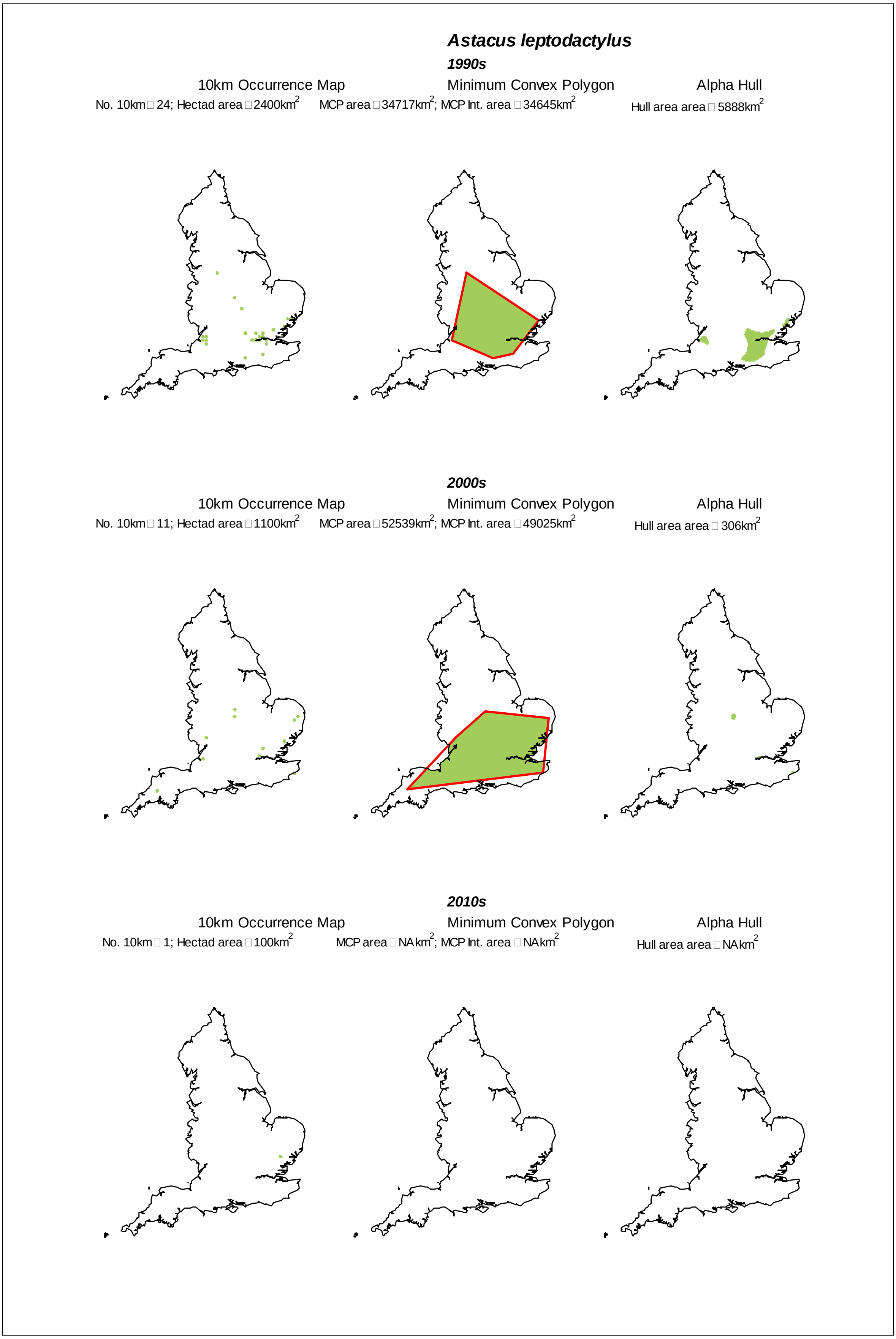


Figure C.3b Change in area of extent for *Astacus leptodactylus* by decade (1990s to 2010s)

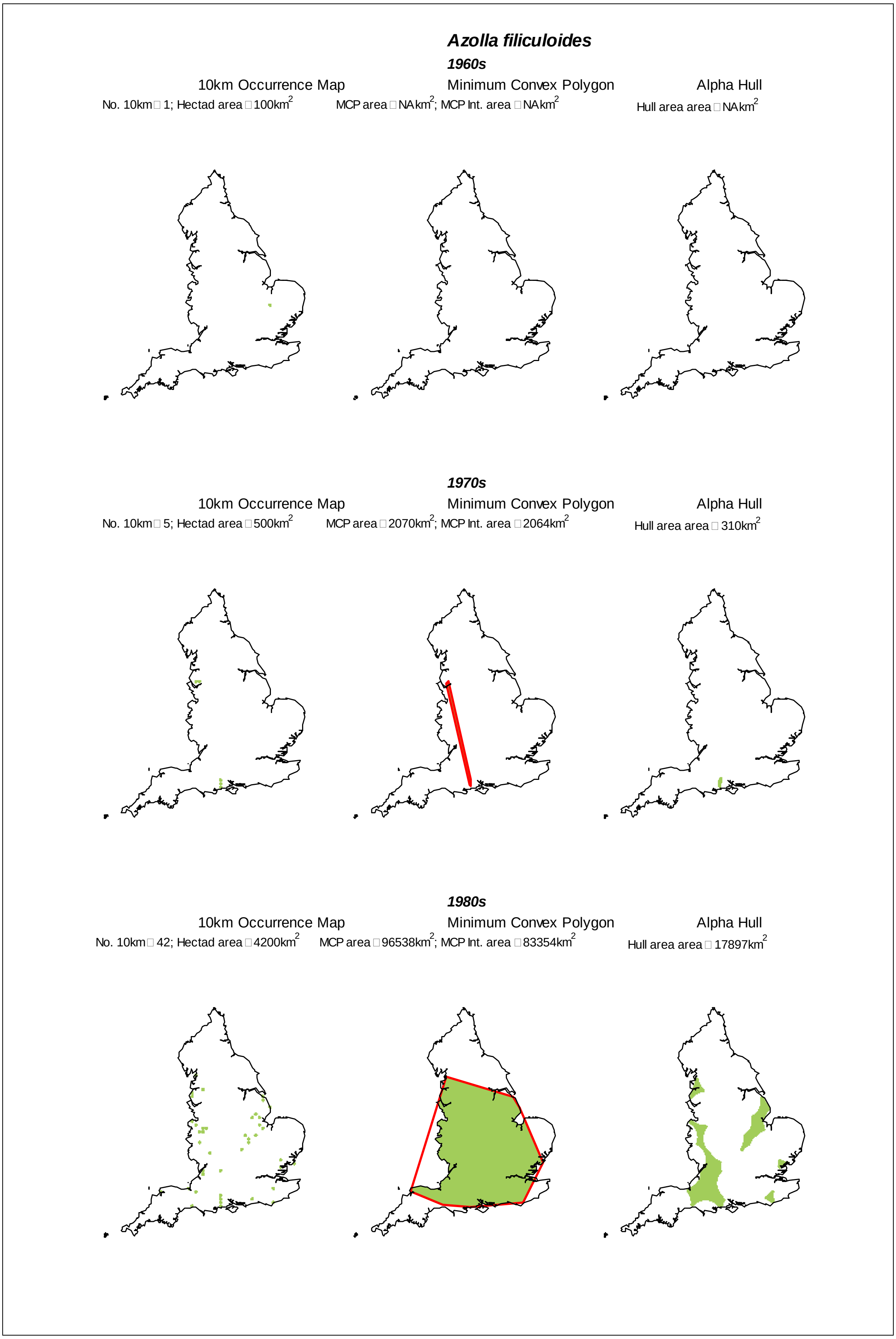


Figure C.4a Change in area of extent for *Azolla filiculoides* by decade (1960s to 1980s)

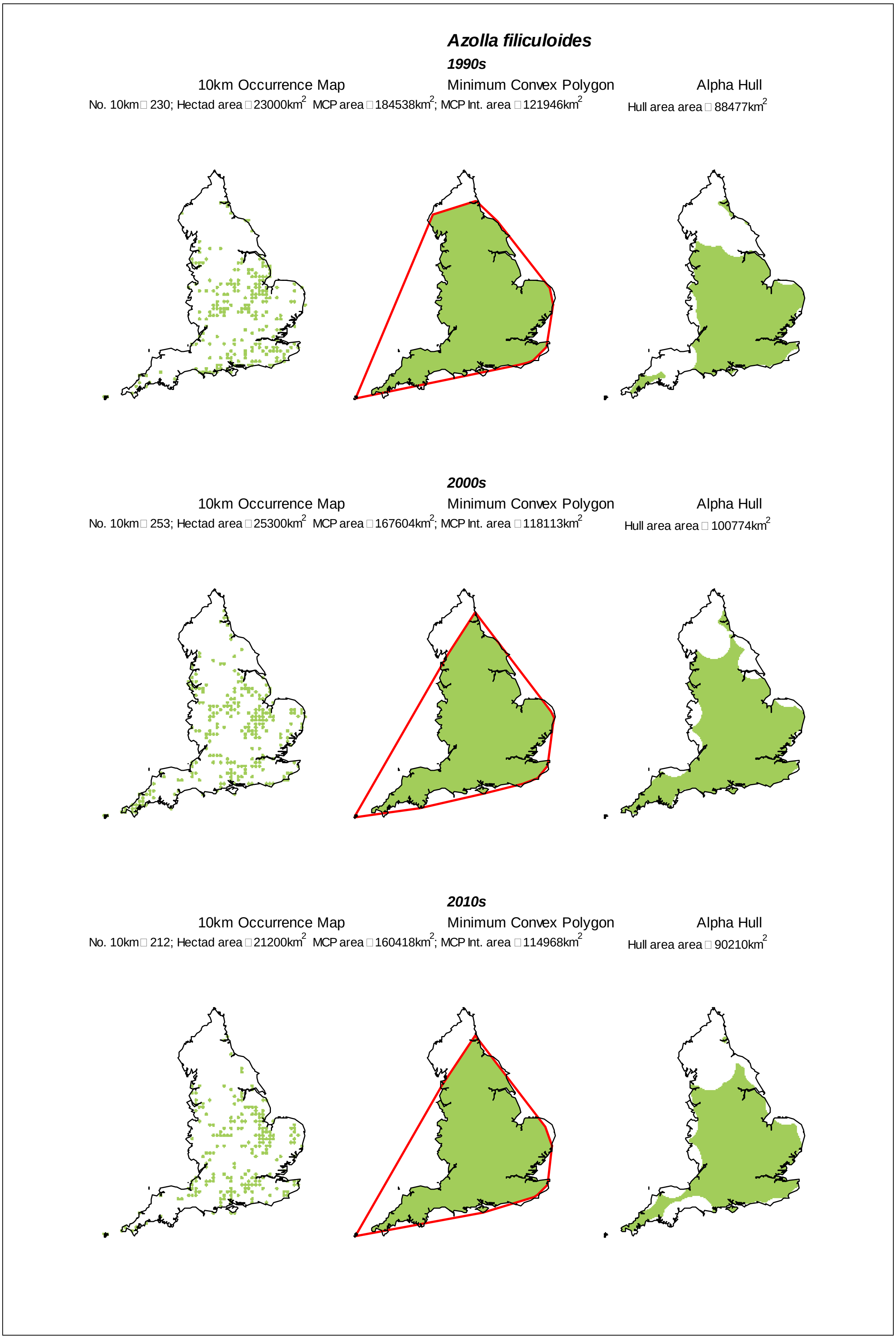


Figure C.4b Change in area of extent for *Azolla filiculoides* by decade (1990s to 2010s)

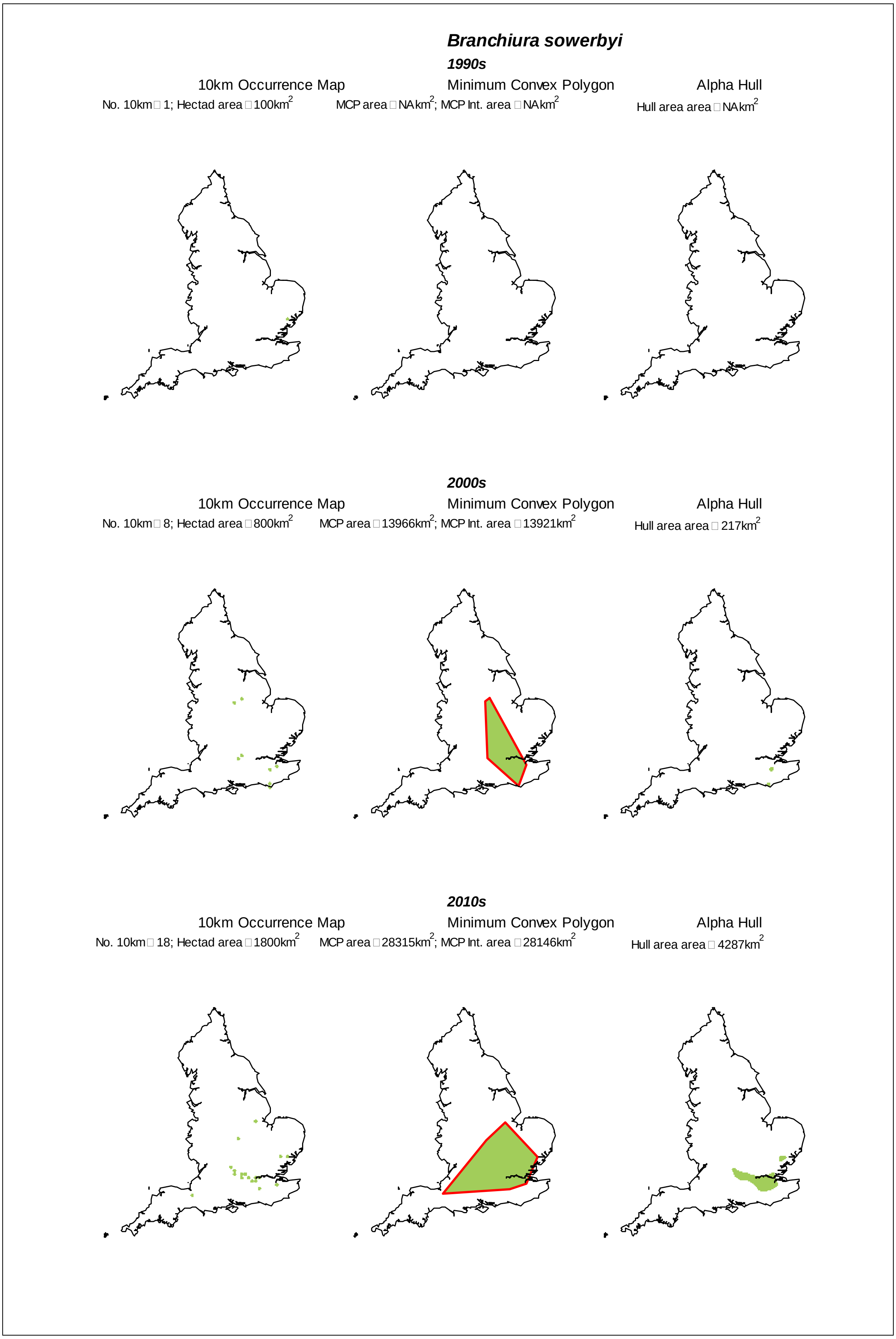


Figure C.5 Change in area of extent for *Branchiura sowerbyi* by decade (1990s to 2010s)

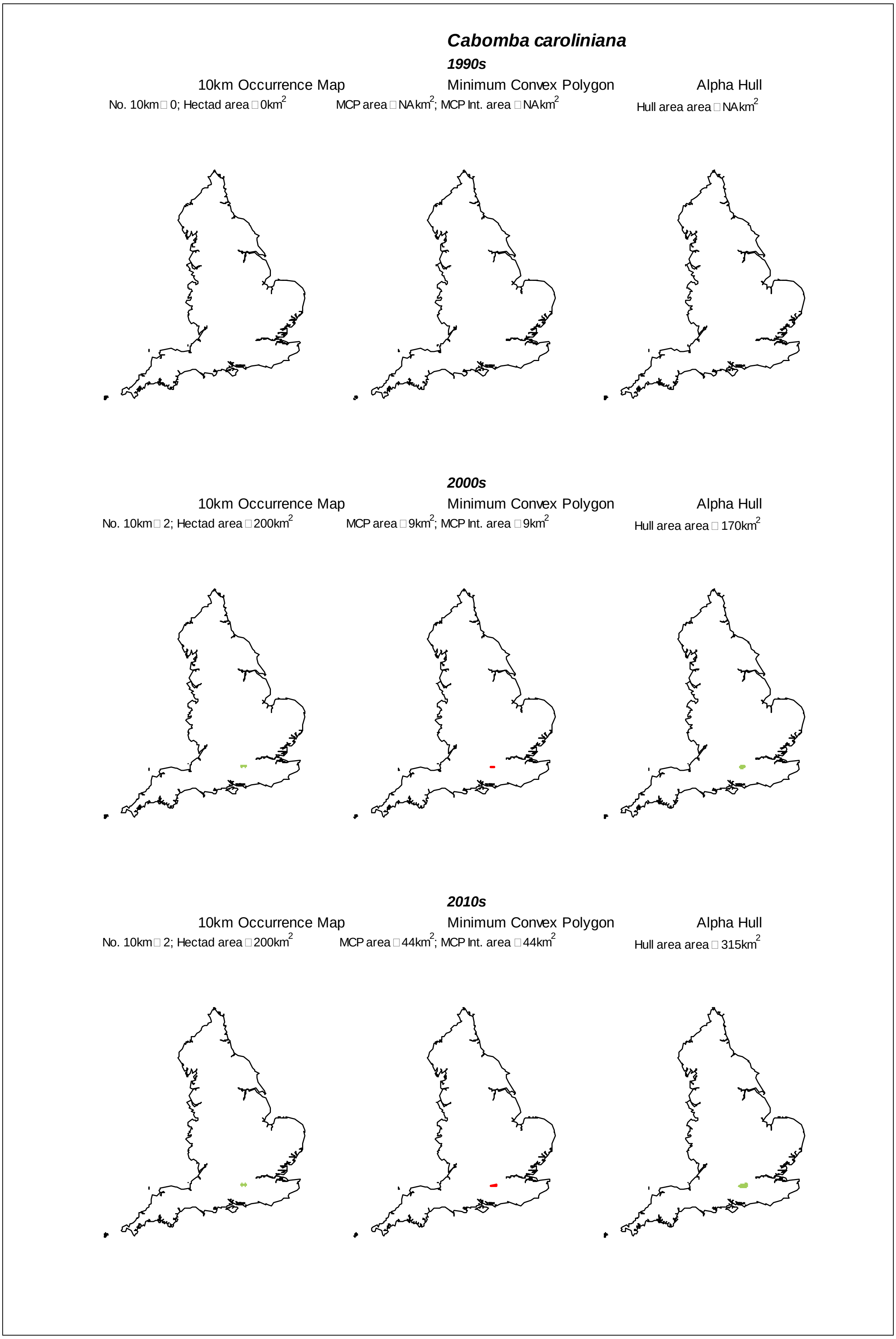


Figure C.6 **Change in area of extent for *Cabomba caroliniana* by decade (1990s to 2010s)**

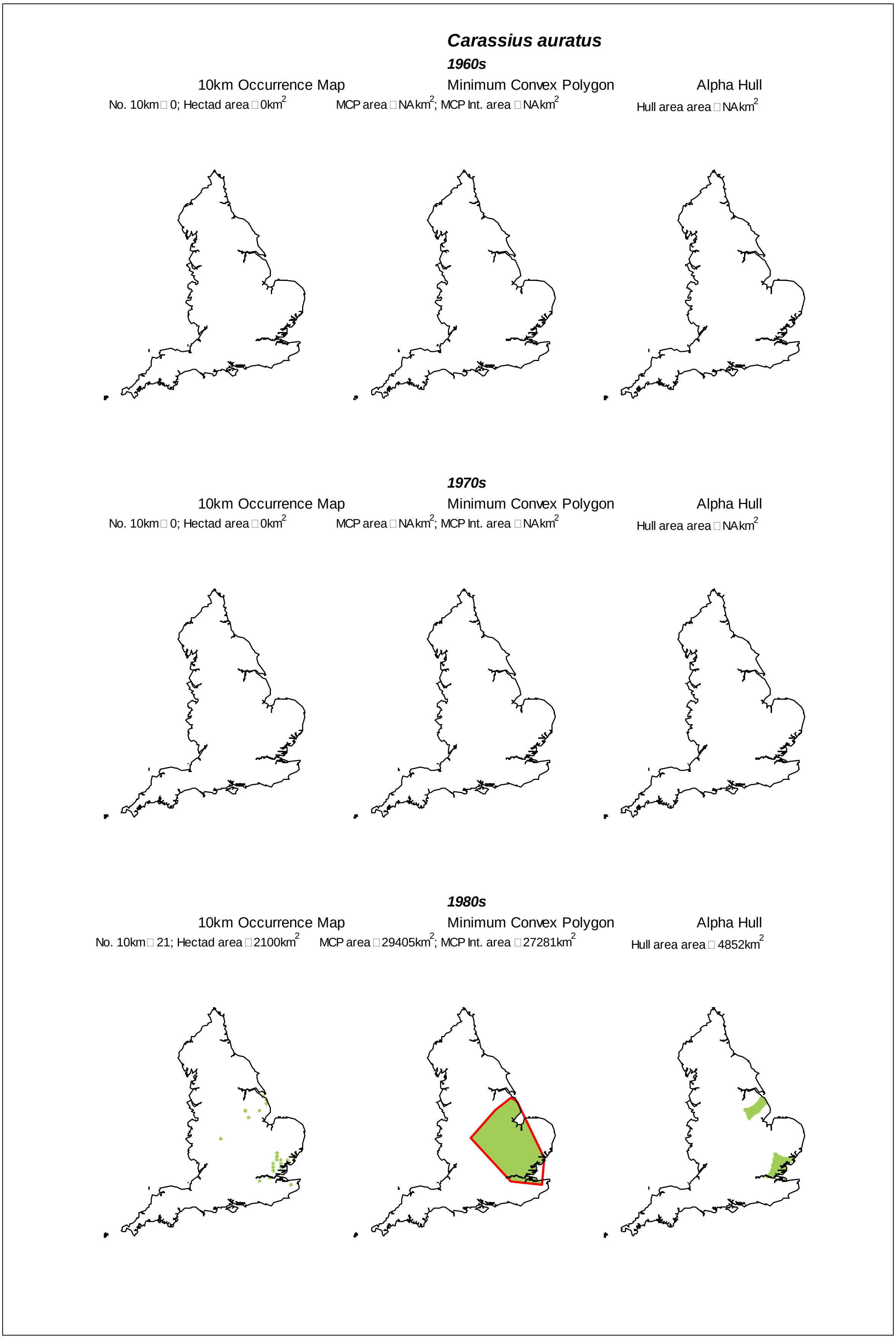


Figure C.7a Change in area of extent for *Carassius auratus* by decade (1960s to 1980s)

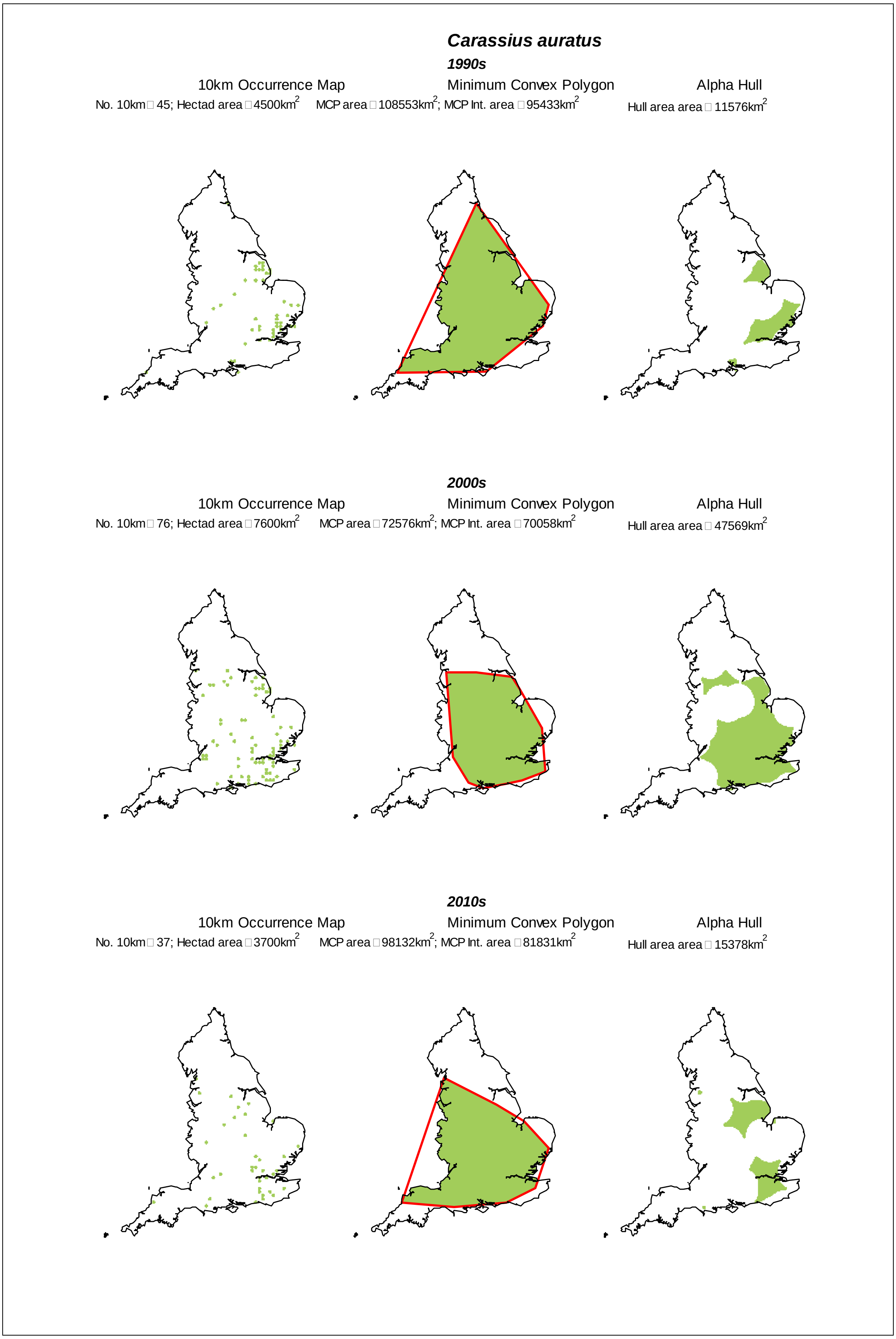


Figure C.7b Change in area of extent for *Carassius auratus* by decade (1990s to 2010s)

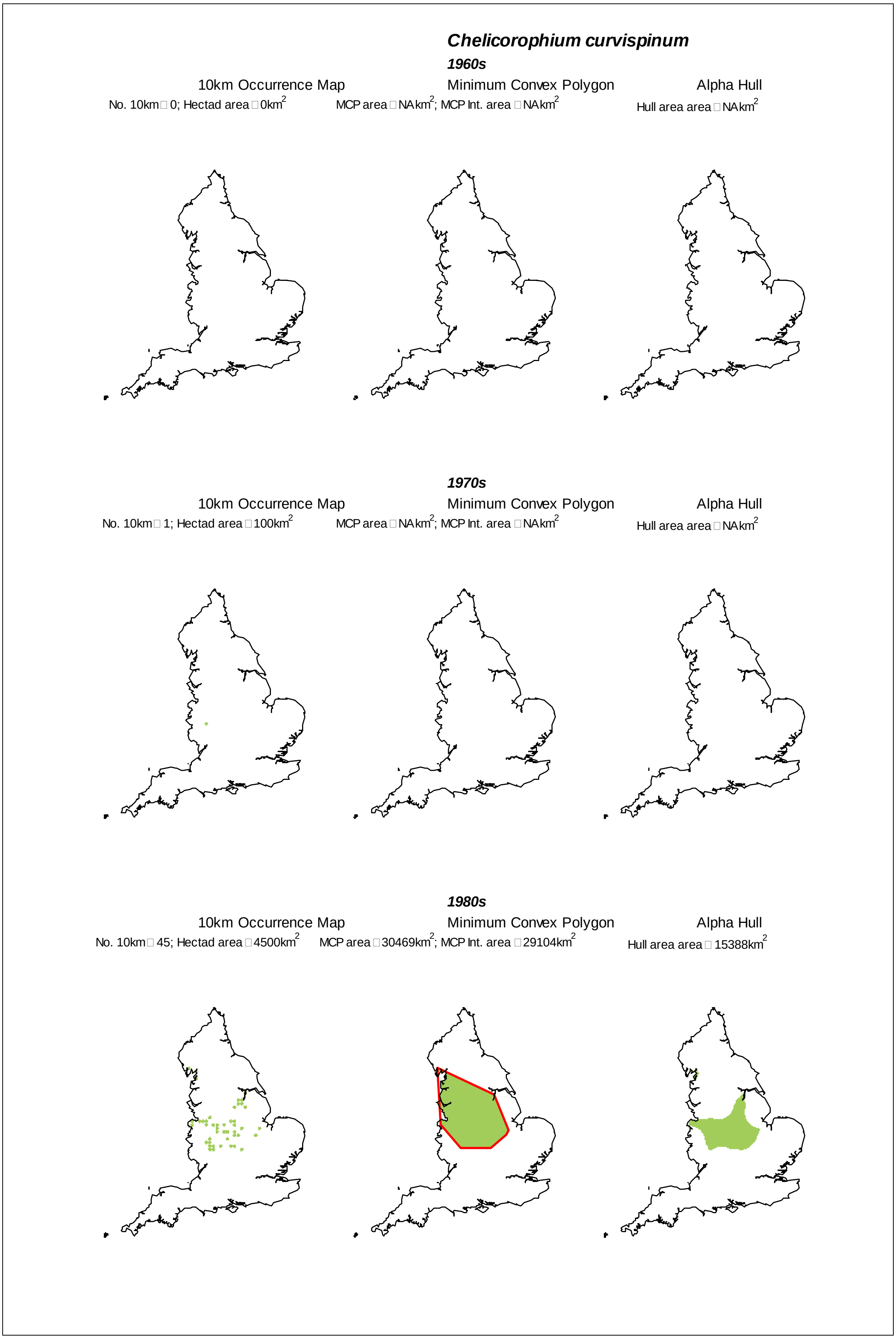


Figure C.8a Change in area of extent for *Chelicorophium curvispinum* by decade (1960s to 1980s)

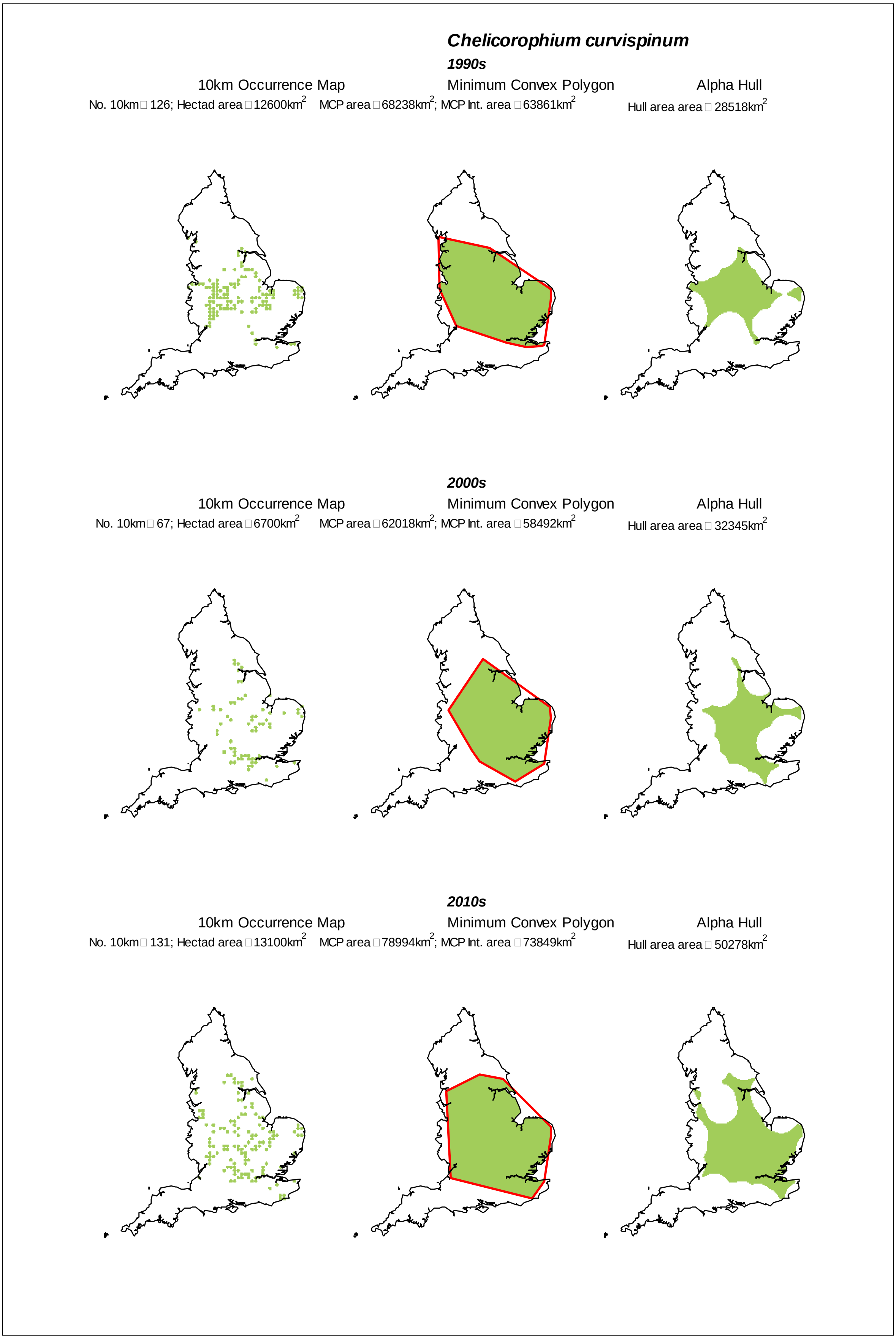


Figure C.8b Change in area of extent for *Chelicorophium curvispinum* by decade (1990s to 2010s)

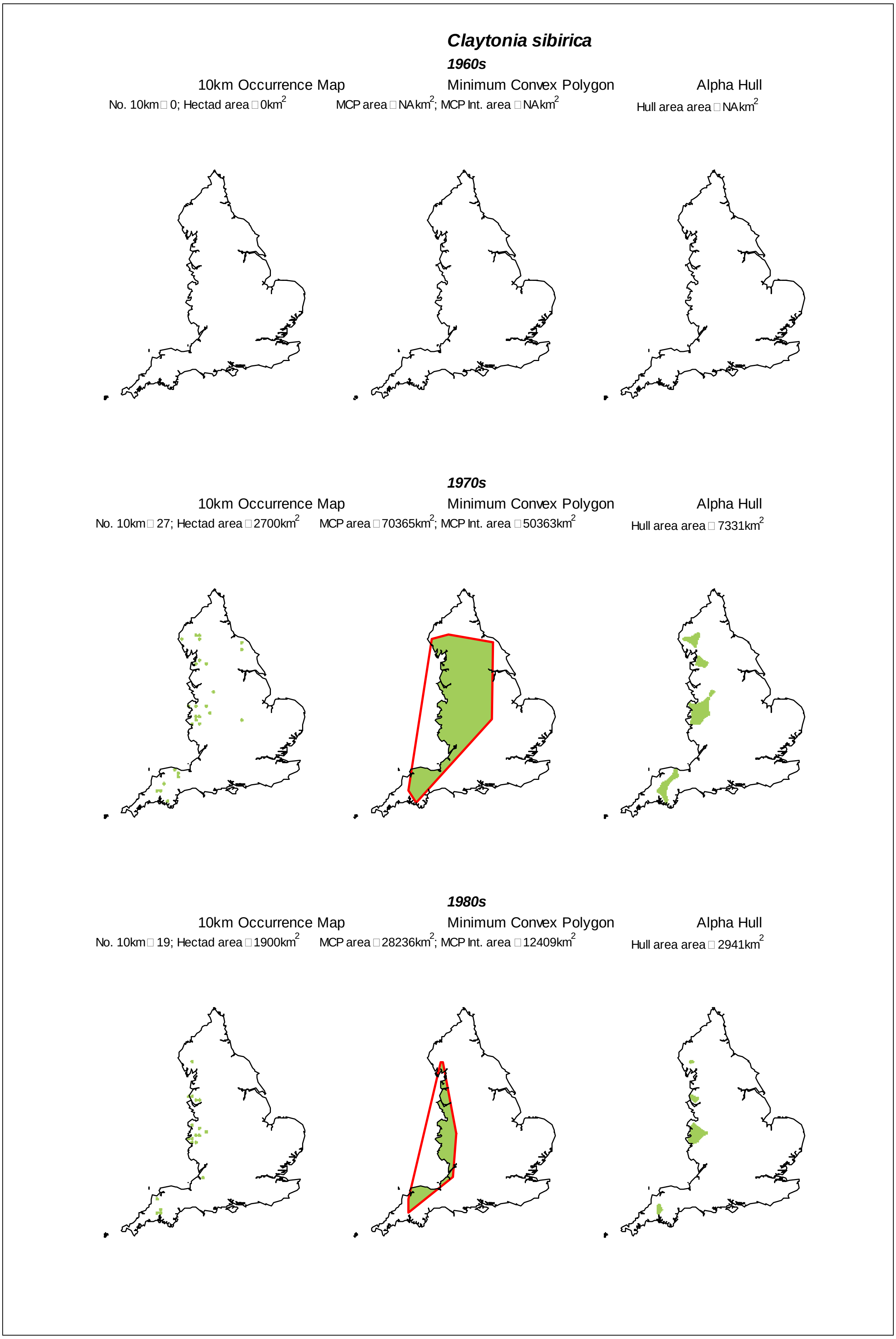


Figure C.9a Change in area of extent for *Claytonia sibirica* by decade (1960s to 1980s)

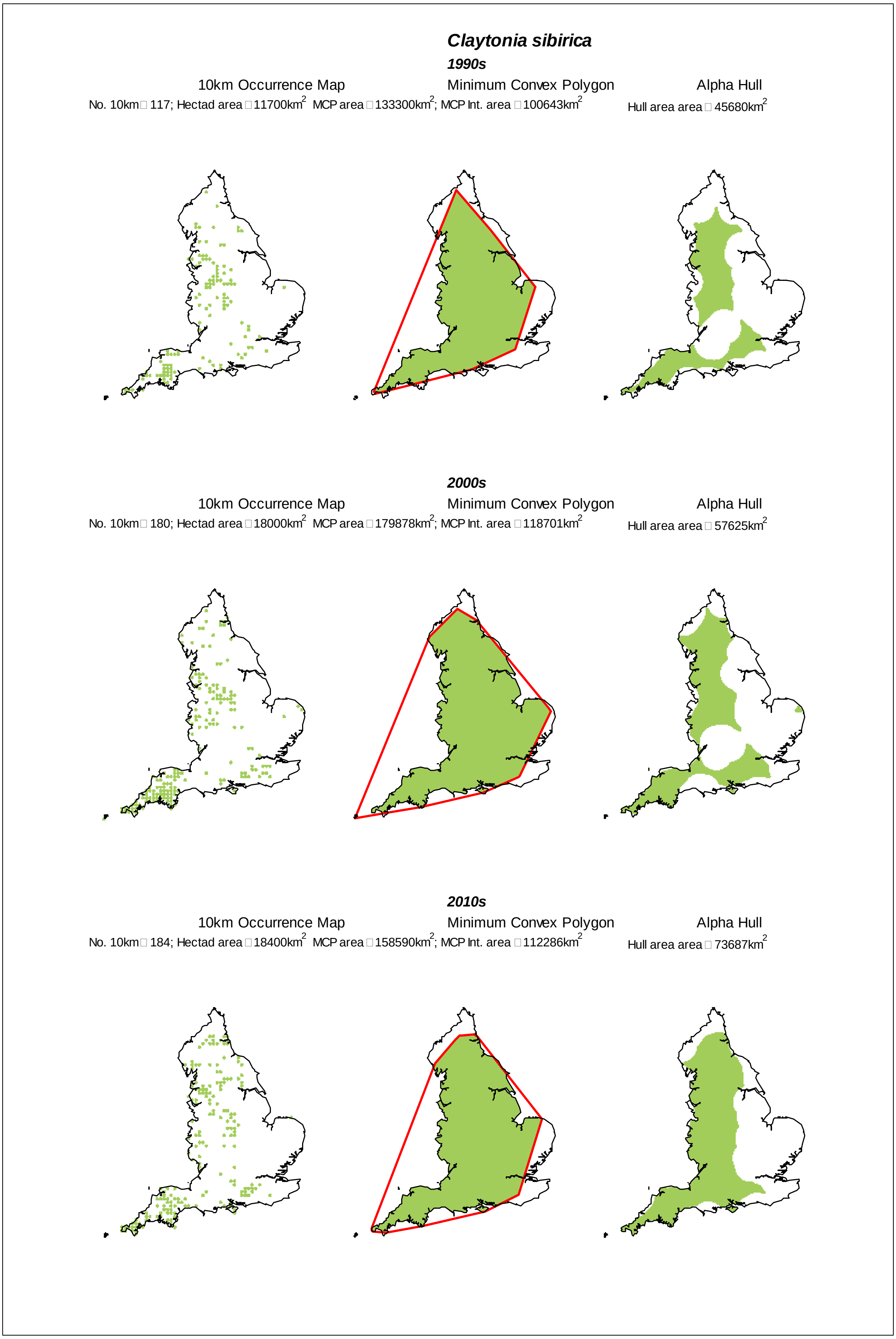


Figure C.9b Change in area of extent for *Claytonia sibirica* by decade (1990s to 2010s)

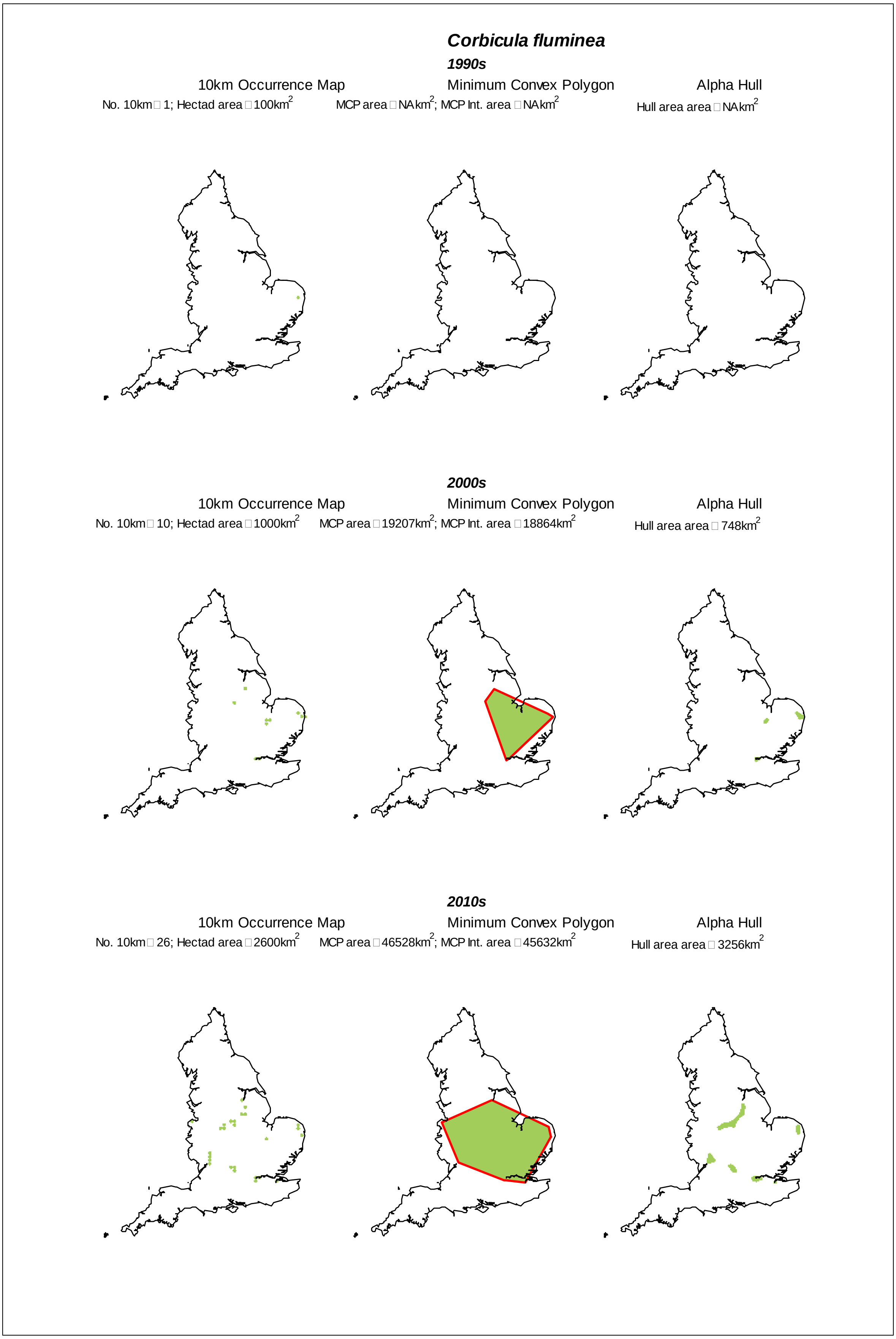


Figure C.10 Change in area of extent for *Corbicula fluminea* by decade (1990s to 2010s)

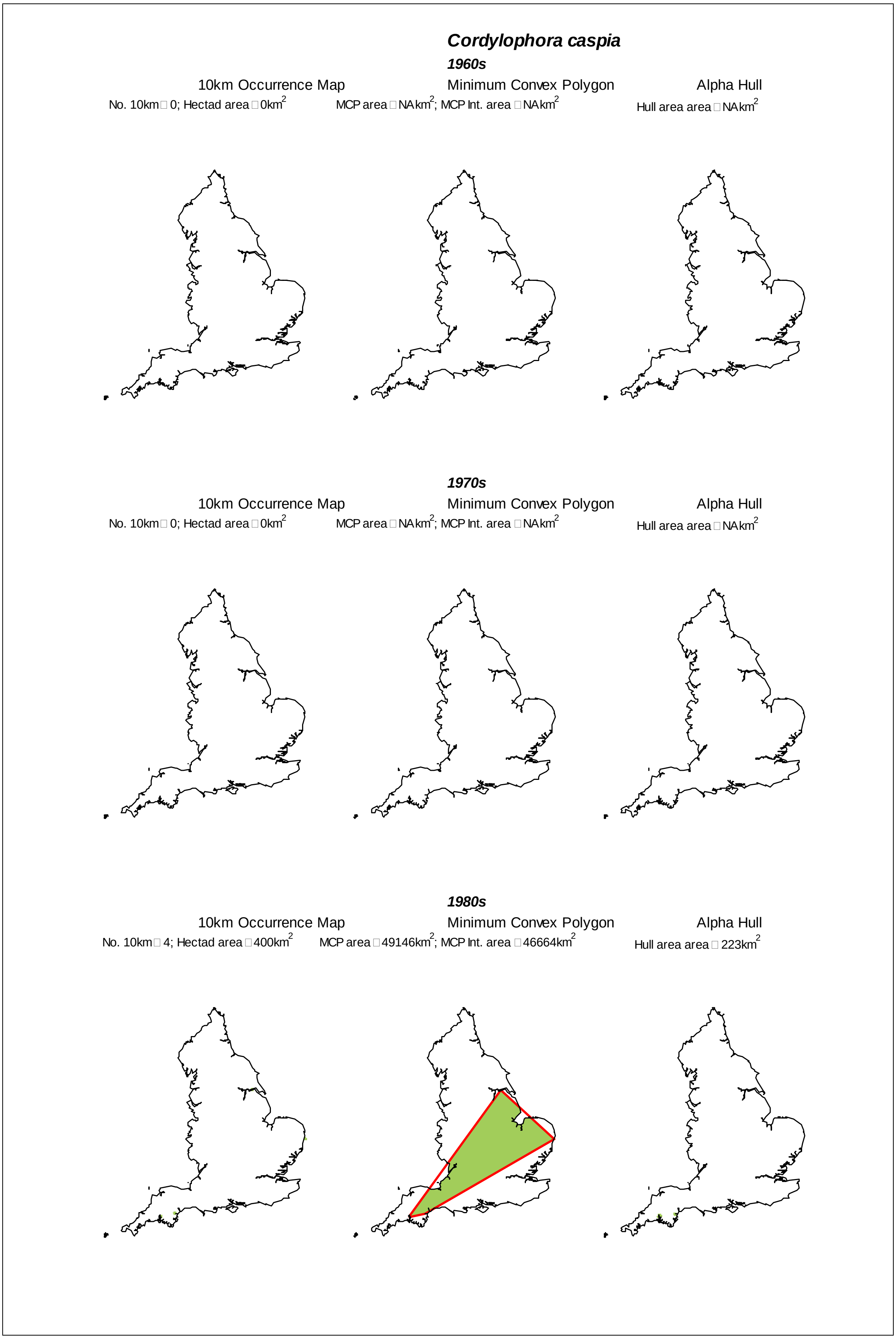


Figure C.11a Change in area of extent for *Cordylophora caspia* by decade (1960s to 1980s)

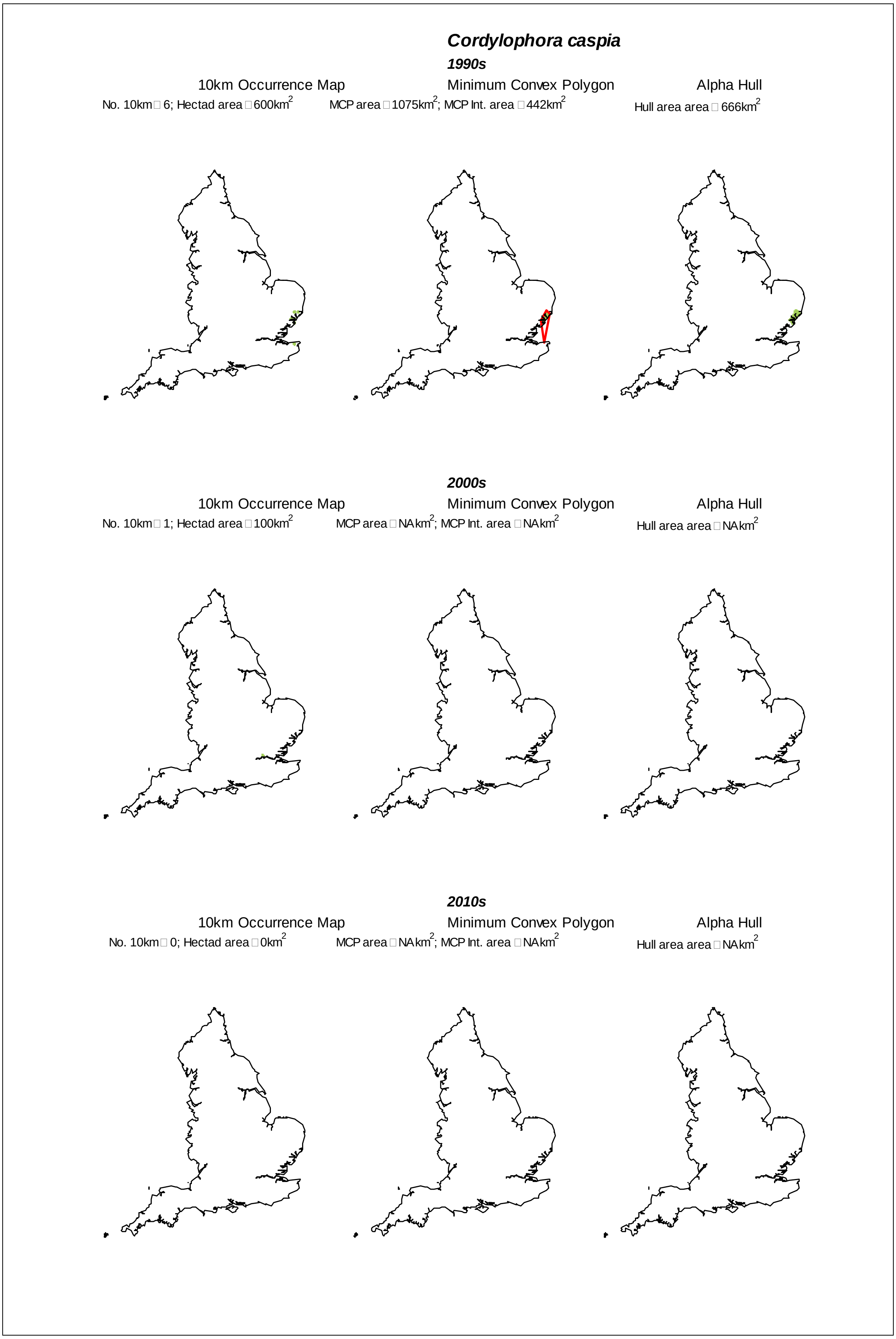


Figure C.11b Change in area of extent for *Cordylophora caspia* by decade (1990s to 2010s)

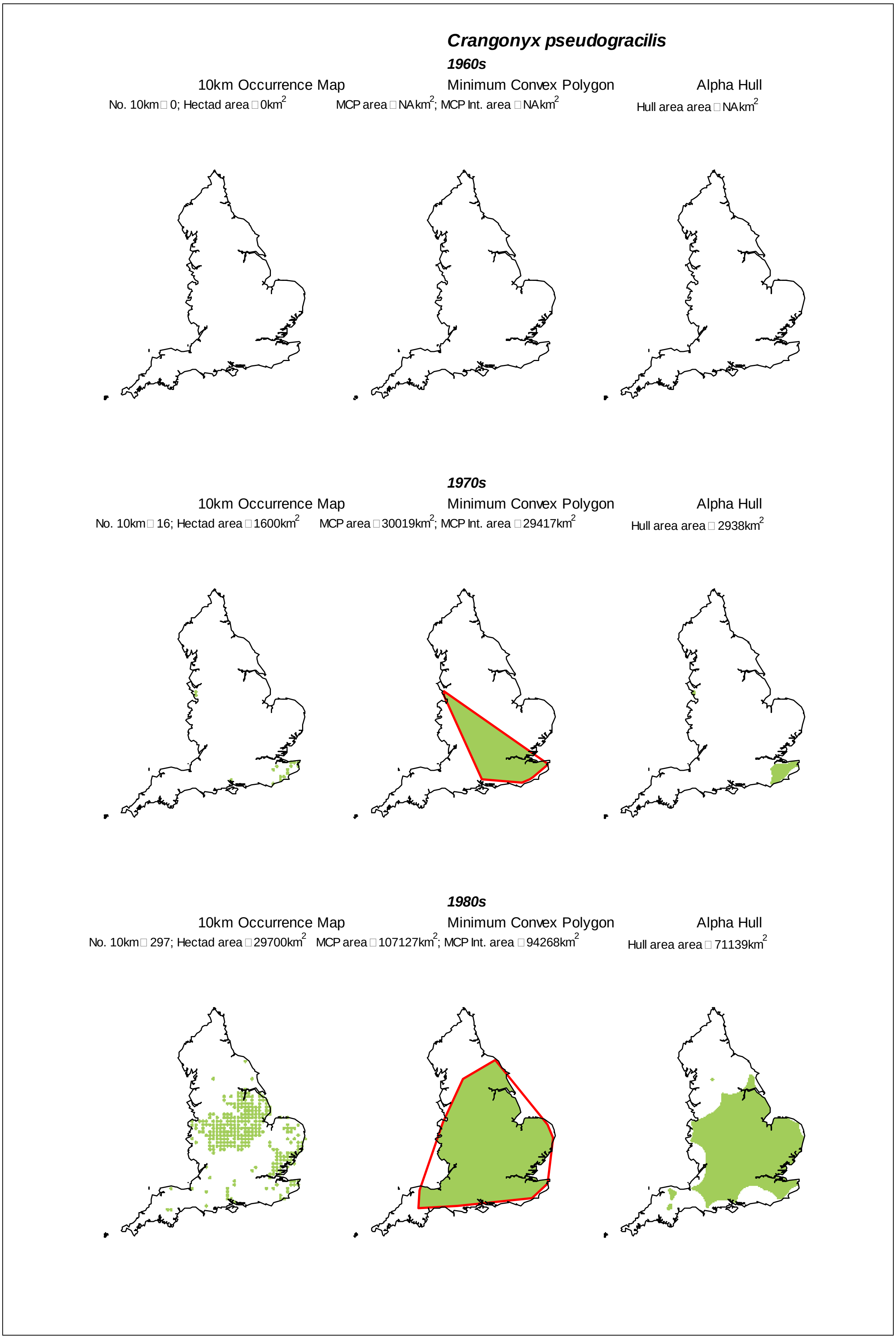


Figure C.12a Change in area of extent for *Crangonyx pseudogracilis* by decade (1960s to 1980s)

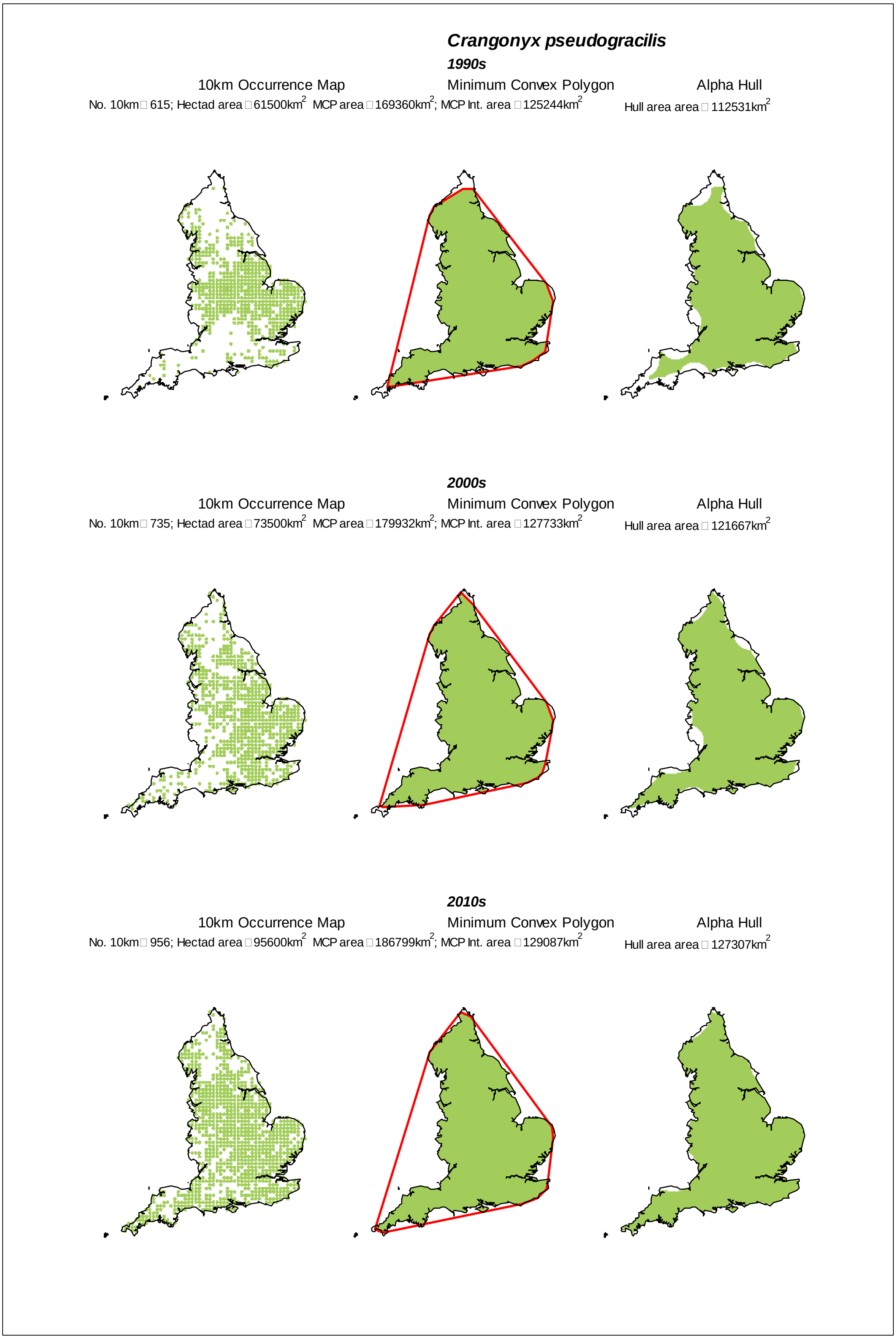


Figure C.12b Change in area of extent for *Crangonyx pseudogracilis* by decade (1990s to 2010s)

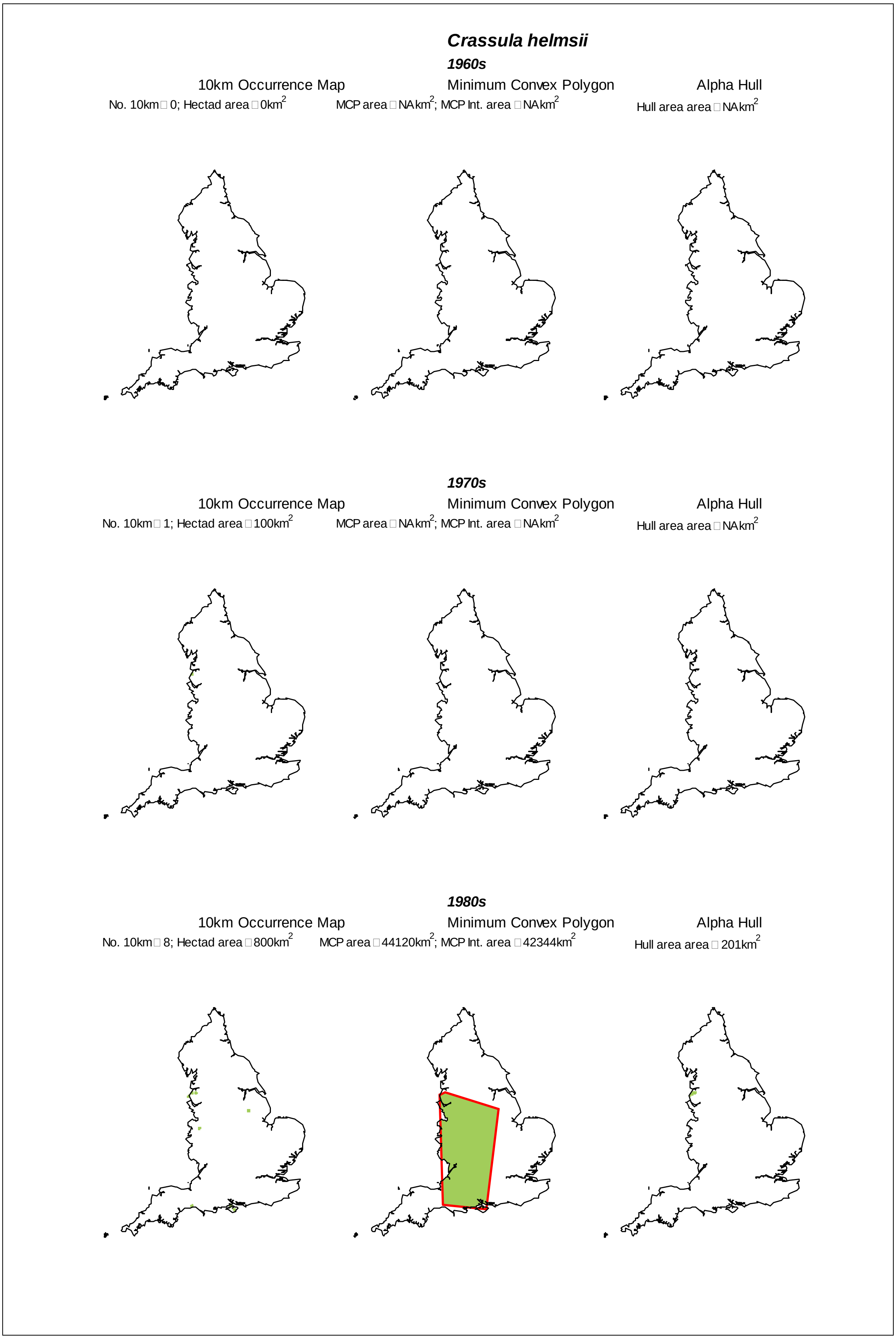


Figure C.13a Change in area of extent for *Crassula helmsii* by decade (1960s to 1980s)

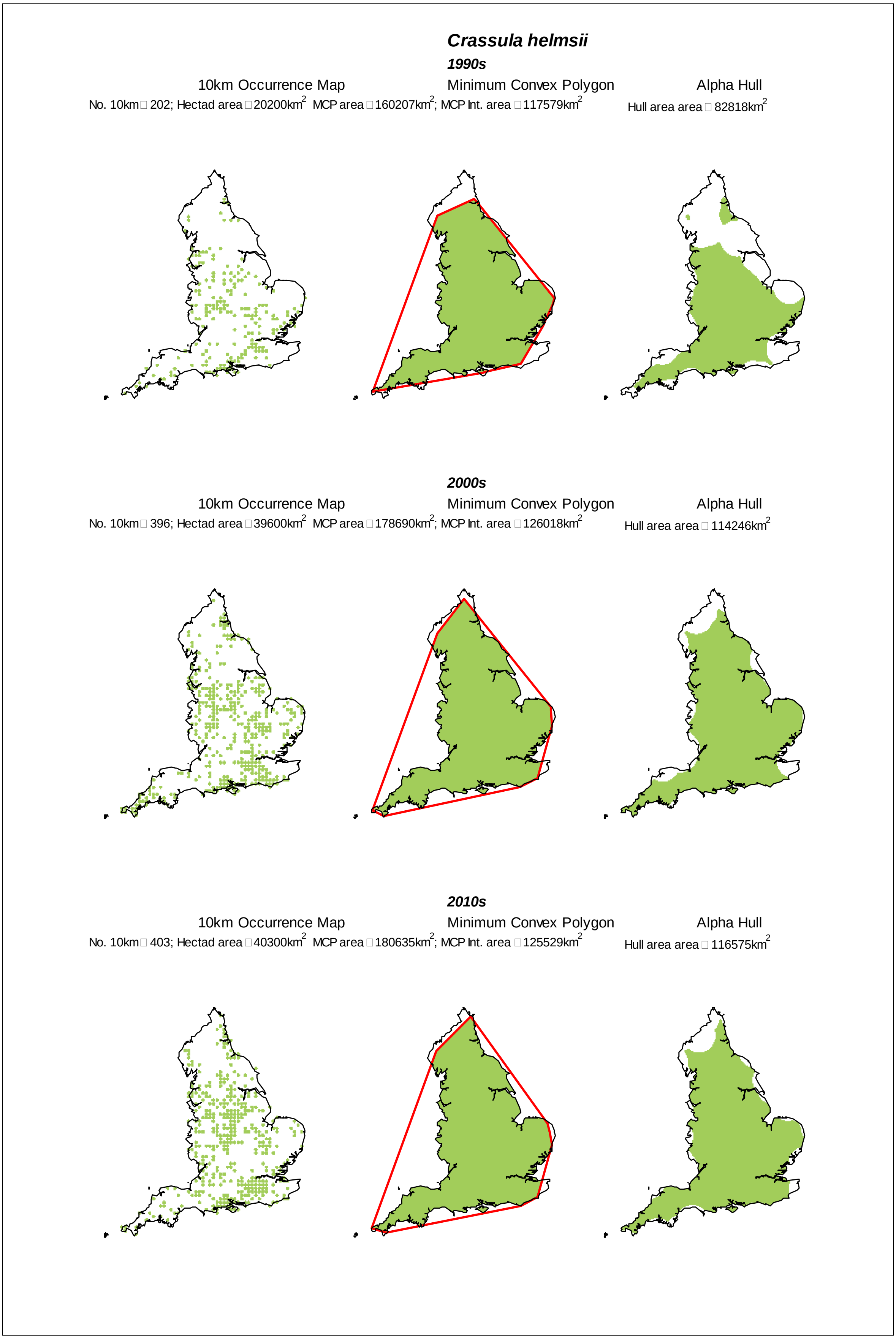


Figure C.13b Change in area of extent for *Crassula helmsii* by decade (1990s to 2010s)

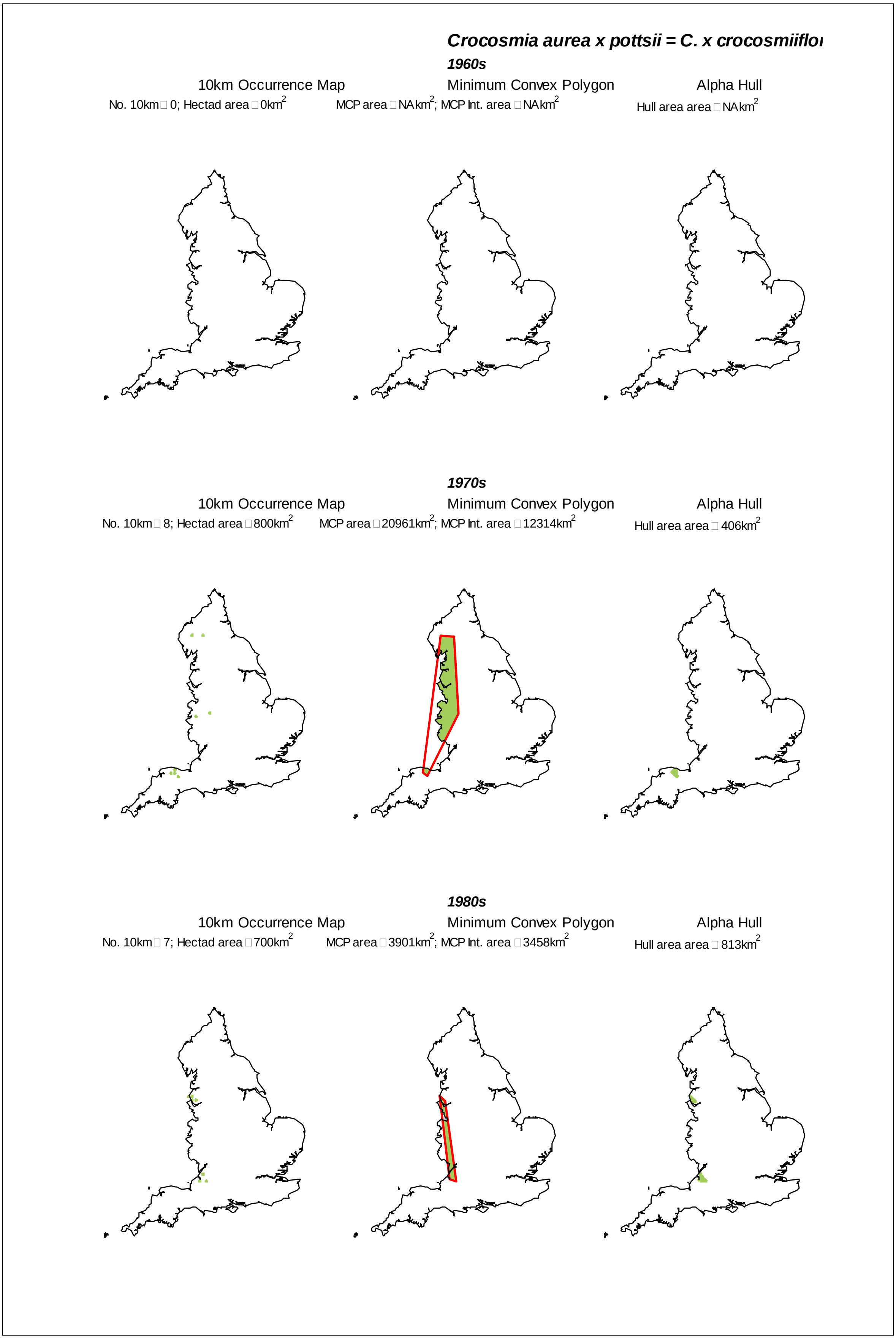


Figure C.14a Change in area of extent for *Crocosmia aurea x pottsii (C. x crocosmiiflora)* by decade (1960s to 1980s)

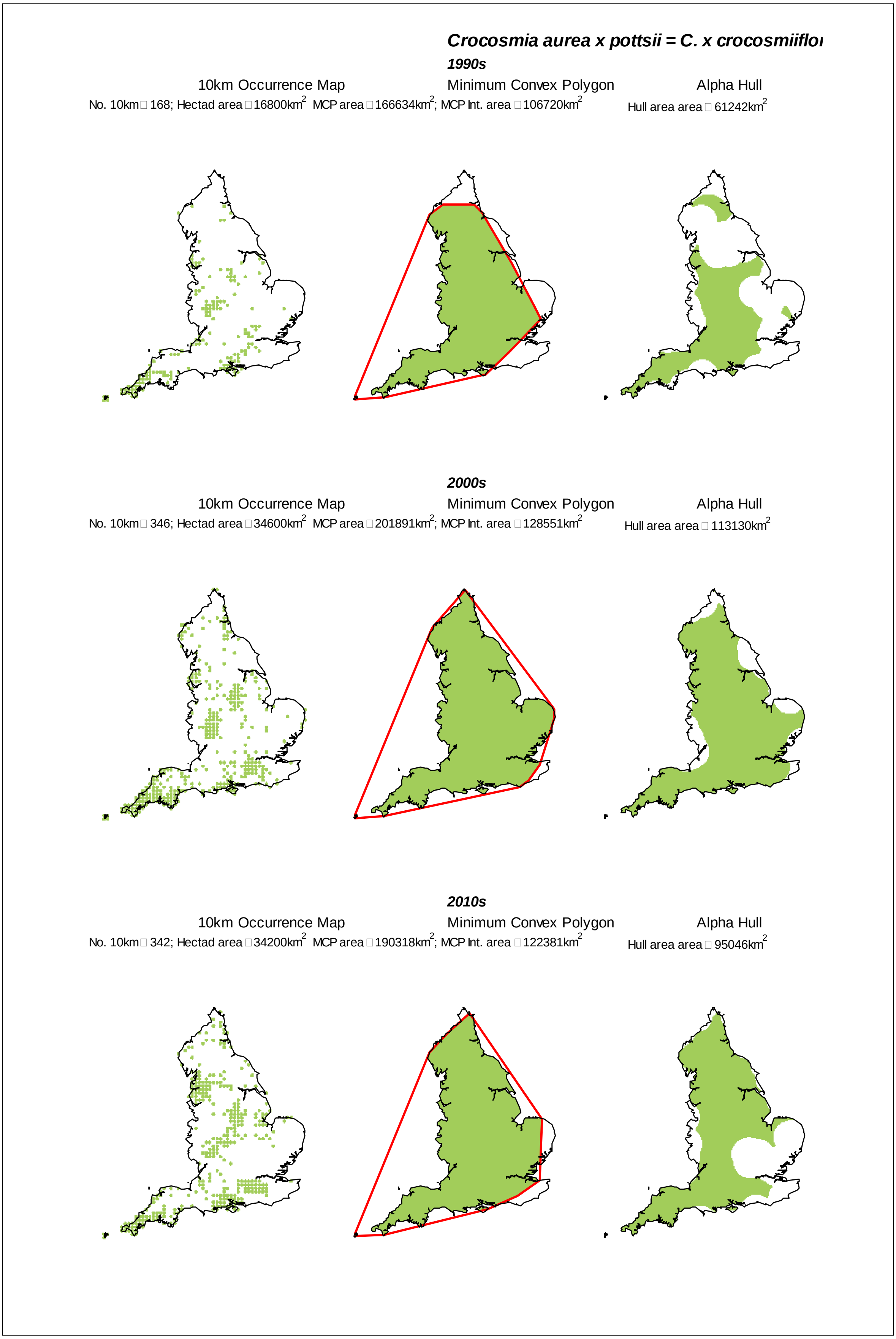


Figure C.14b Change in area of extent for *Crocosmia aurea x pottsii (C. x crocosmiiflora)* by decade (1990s to 2010s)

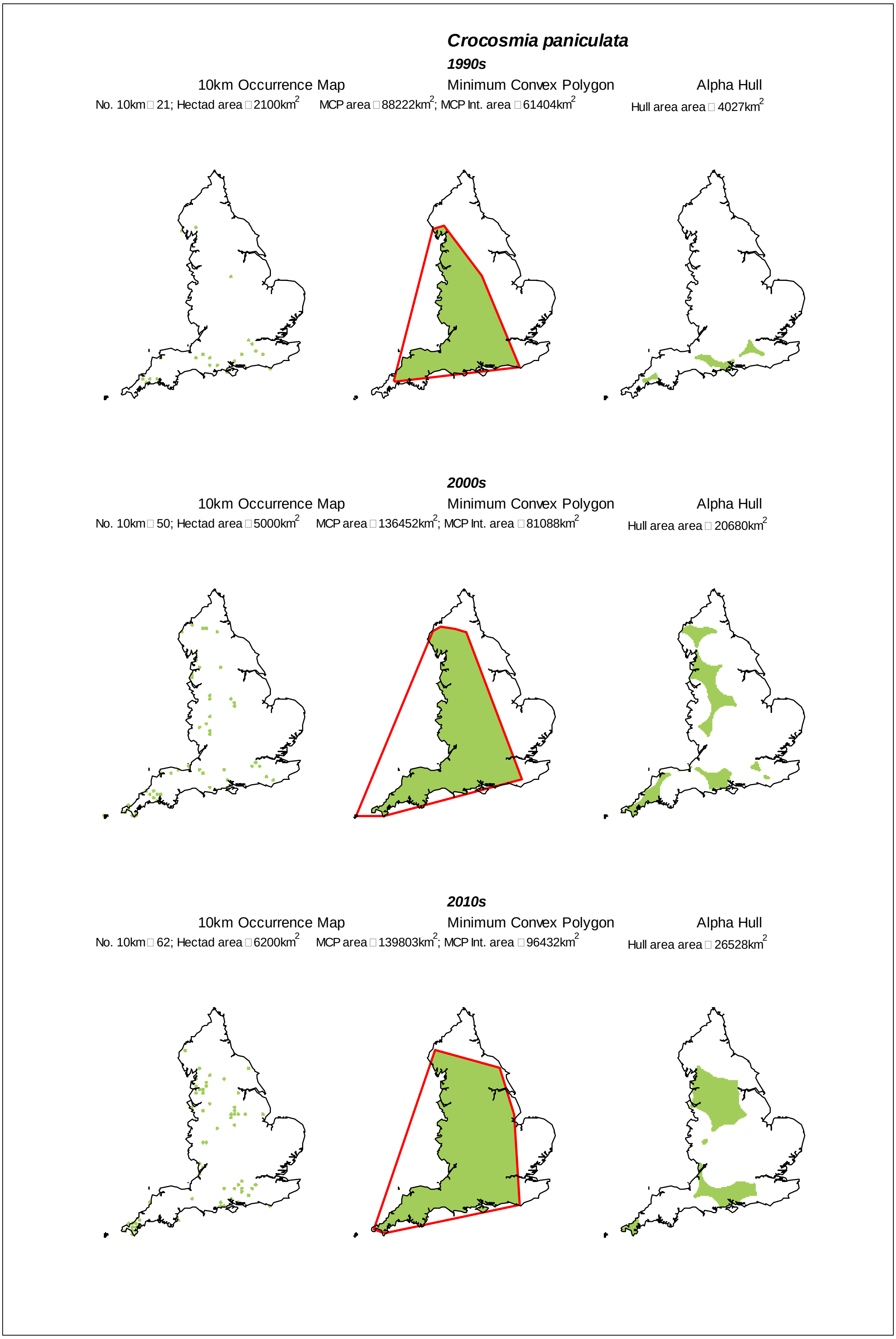


Figure C.15 Change in area of extent for *Crocosmia paniculata* by decade (1990s to 2010s)

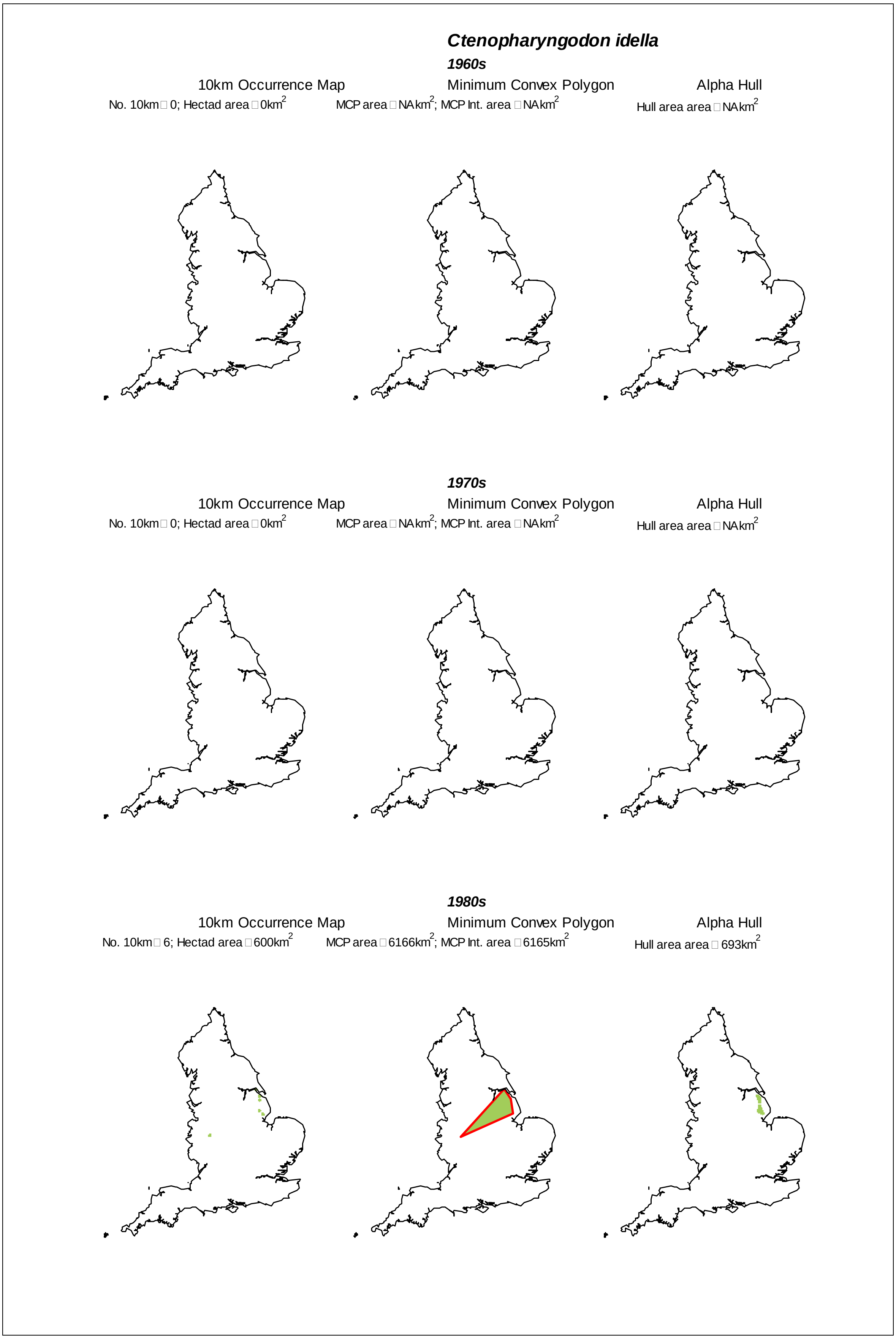


Figure C.16a Change in area of extent for *Ctenopharyngodon idella* by decade (1960s to 1980s)

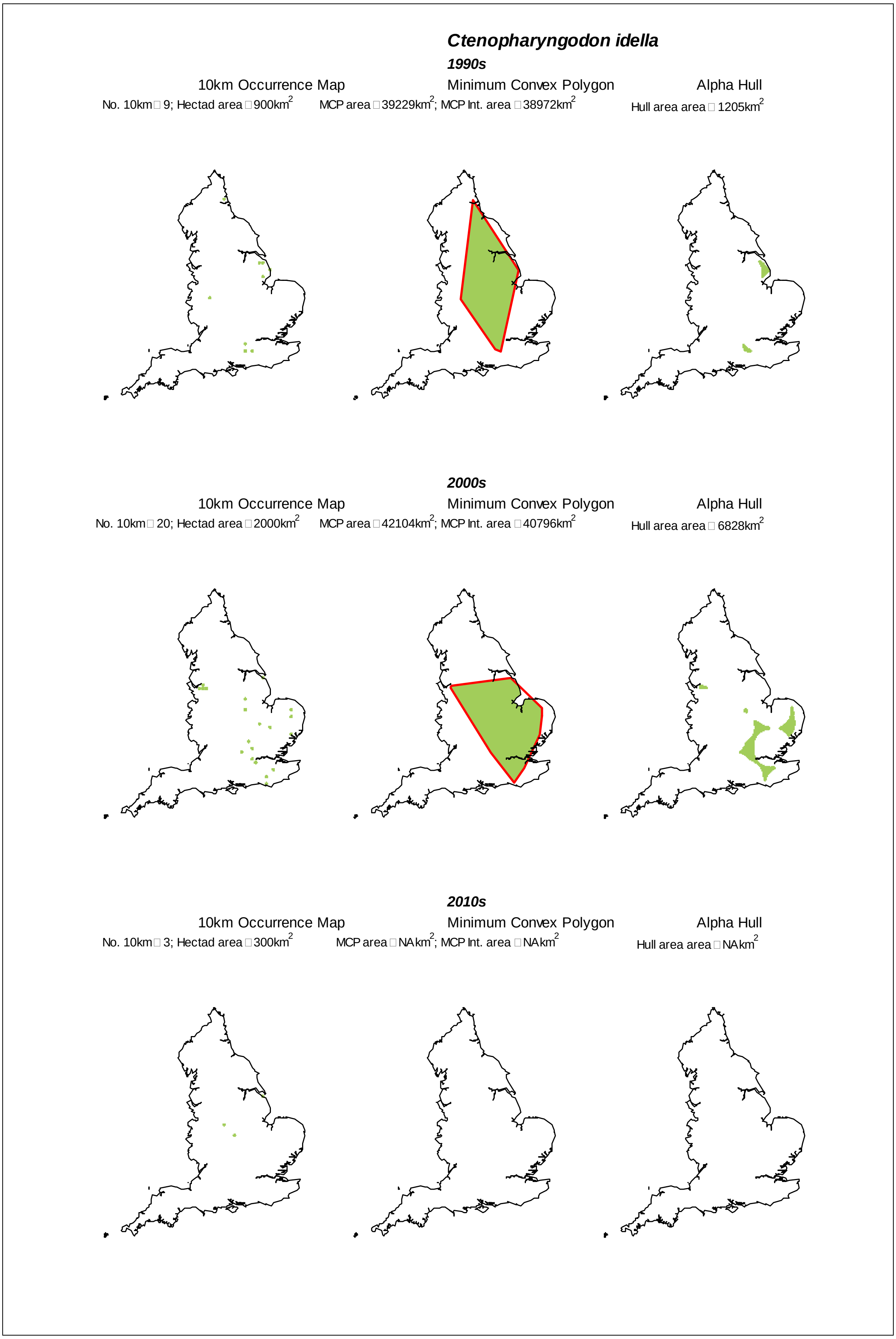


Figure C.16b Change in area of extent for *Ctenopharyngodon idella* by decade (1990s to 2010s)

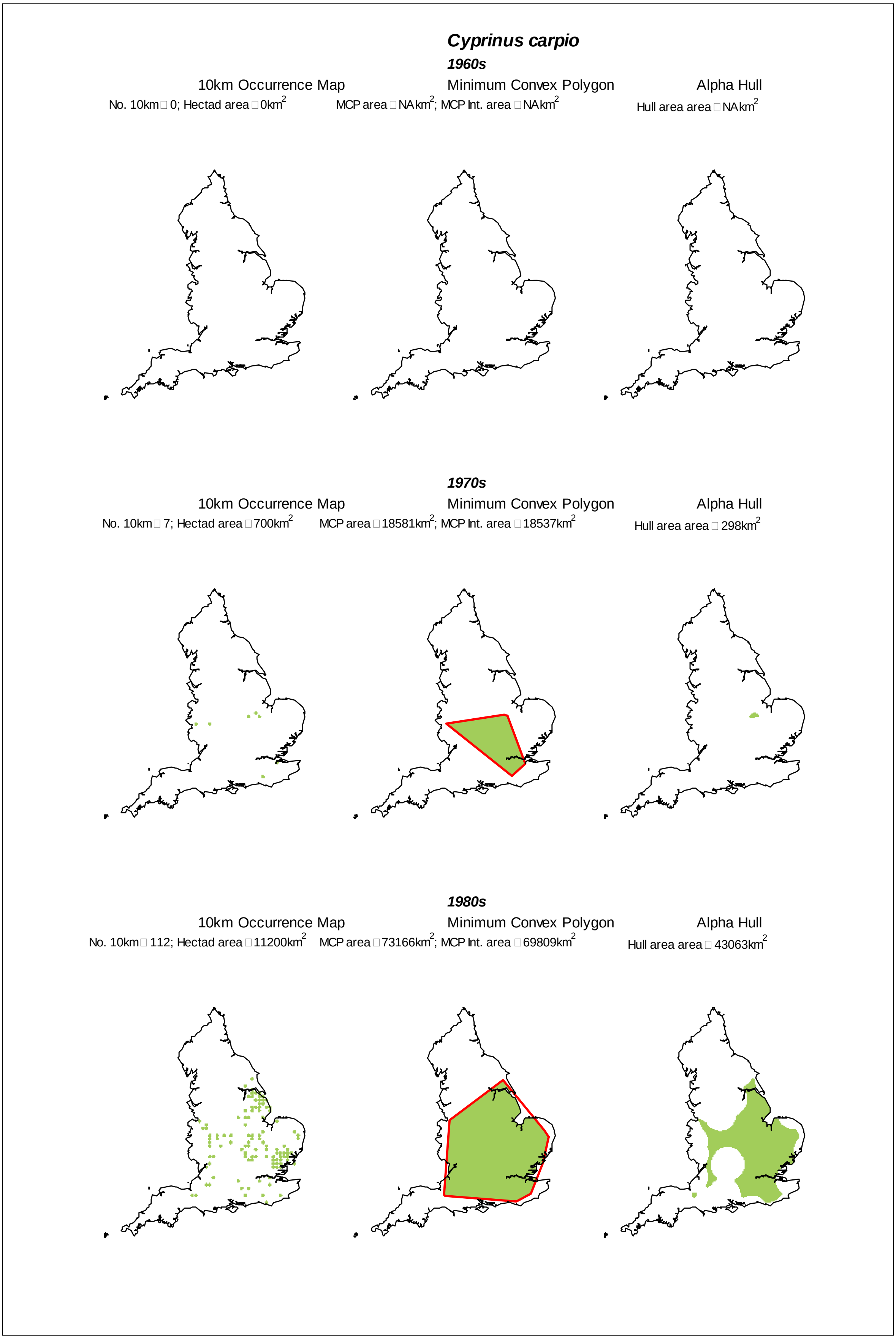


Figure C.17a Change in area of extent for *Cyprinus carpio* by decade (1960s to 1980s)

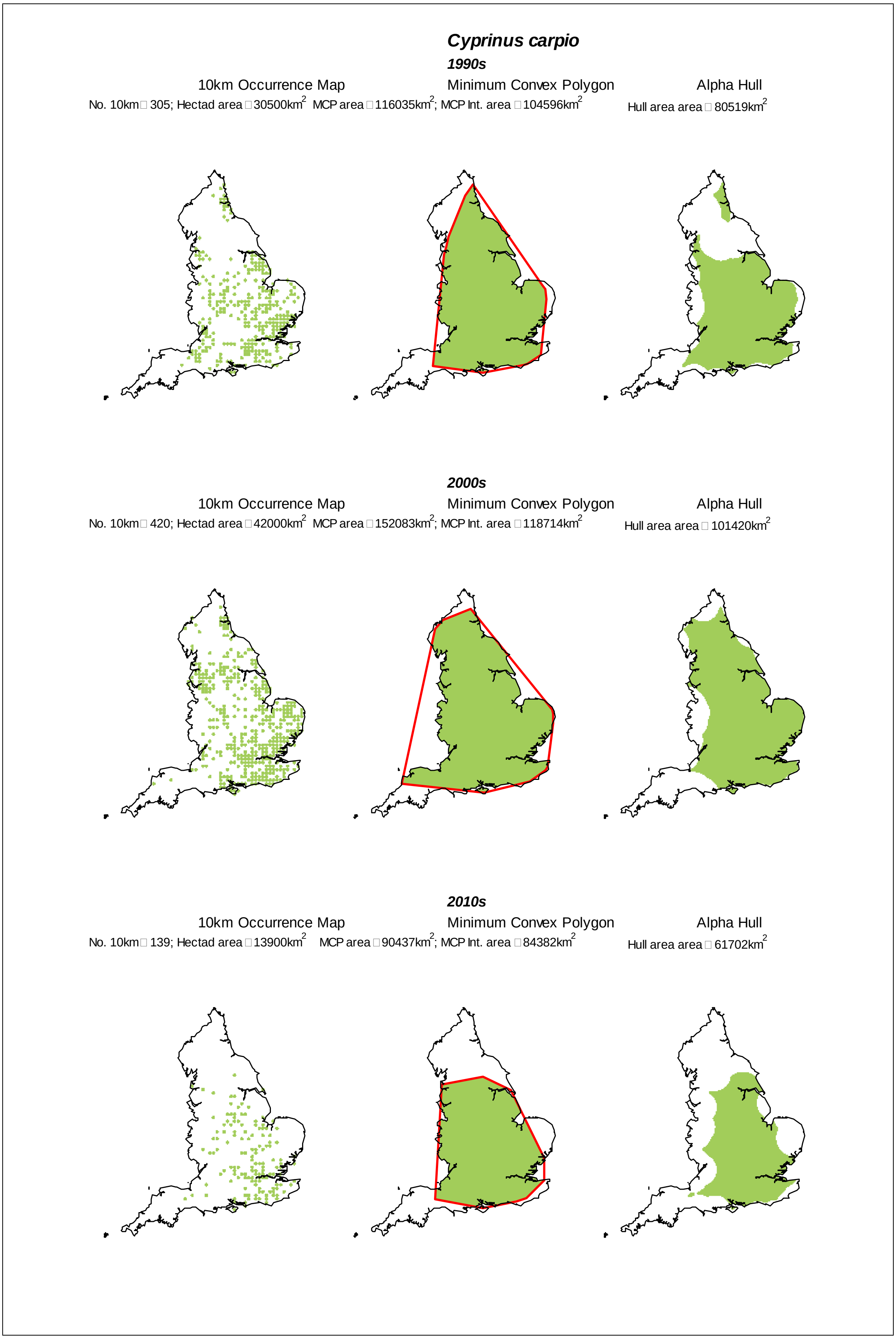


Figure C.17b Change in area of extent for *Cyprinus carpio* by decade (1990s to 2010s)

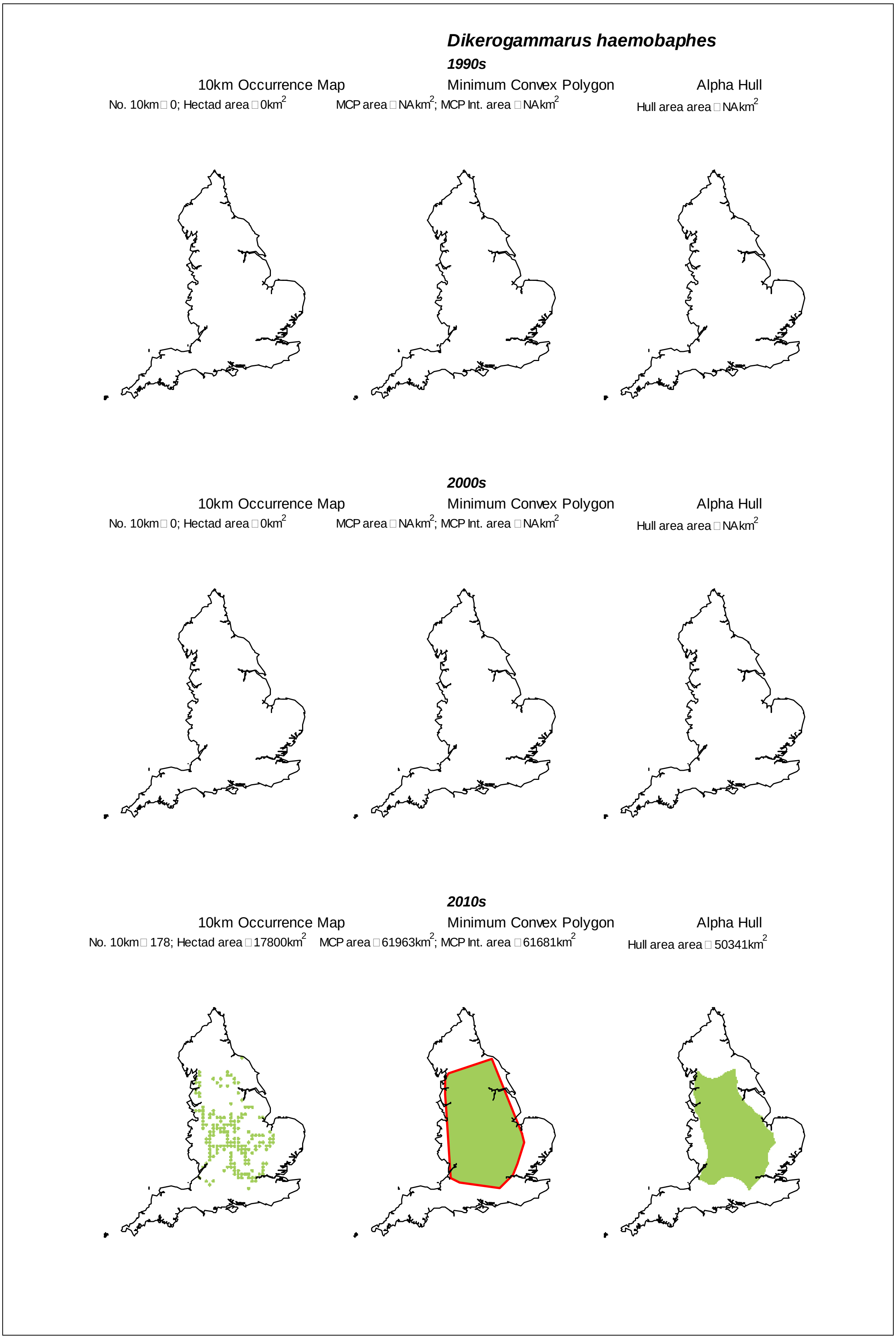


Figure C.18 Change in area of extent for *Dikerogammarus haemobaphes* by decade (1990s to 2010s)

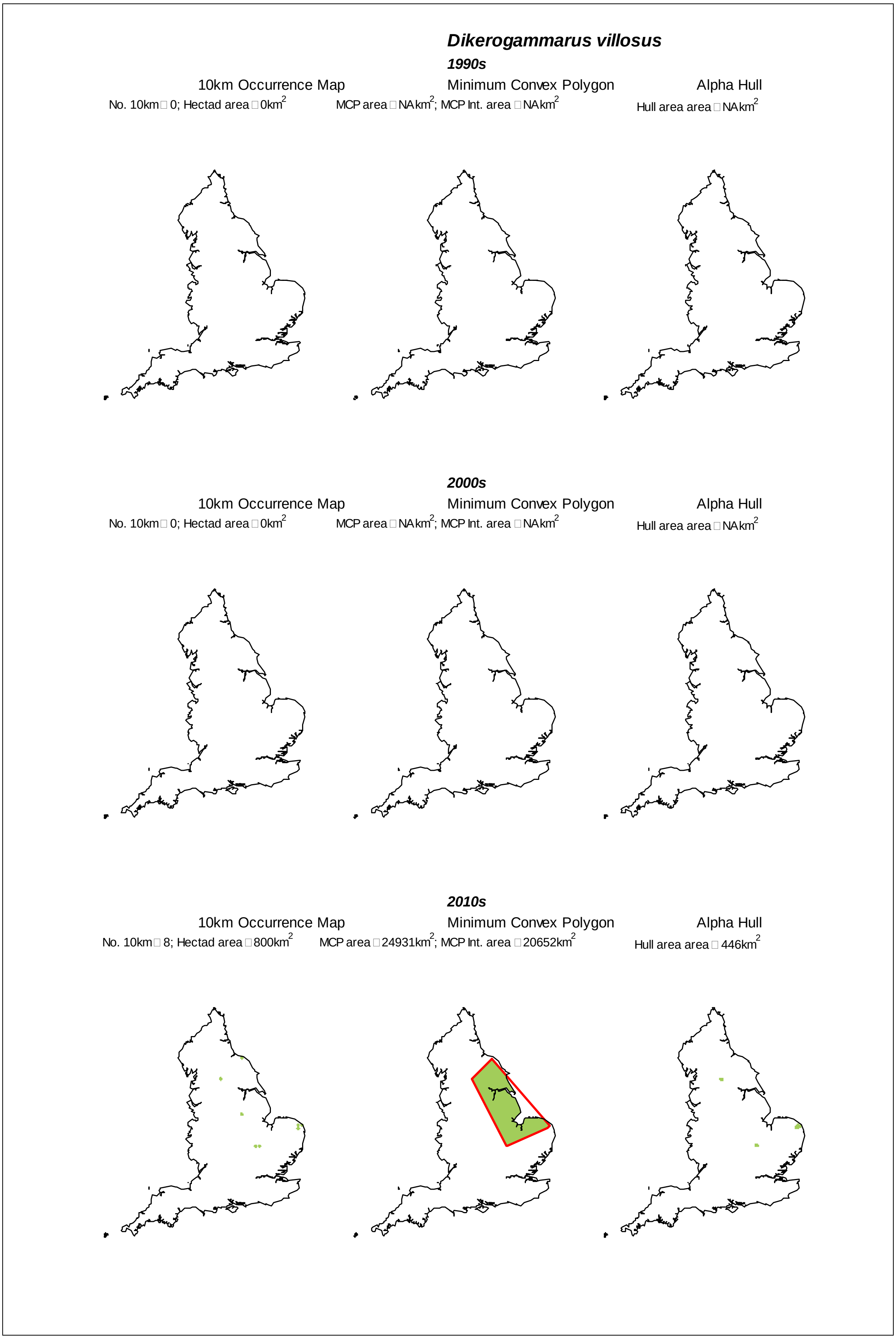


Figure C.19 Change in area of extent for *Dikerogammarus villosus* by decade (1990s to 2010s)

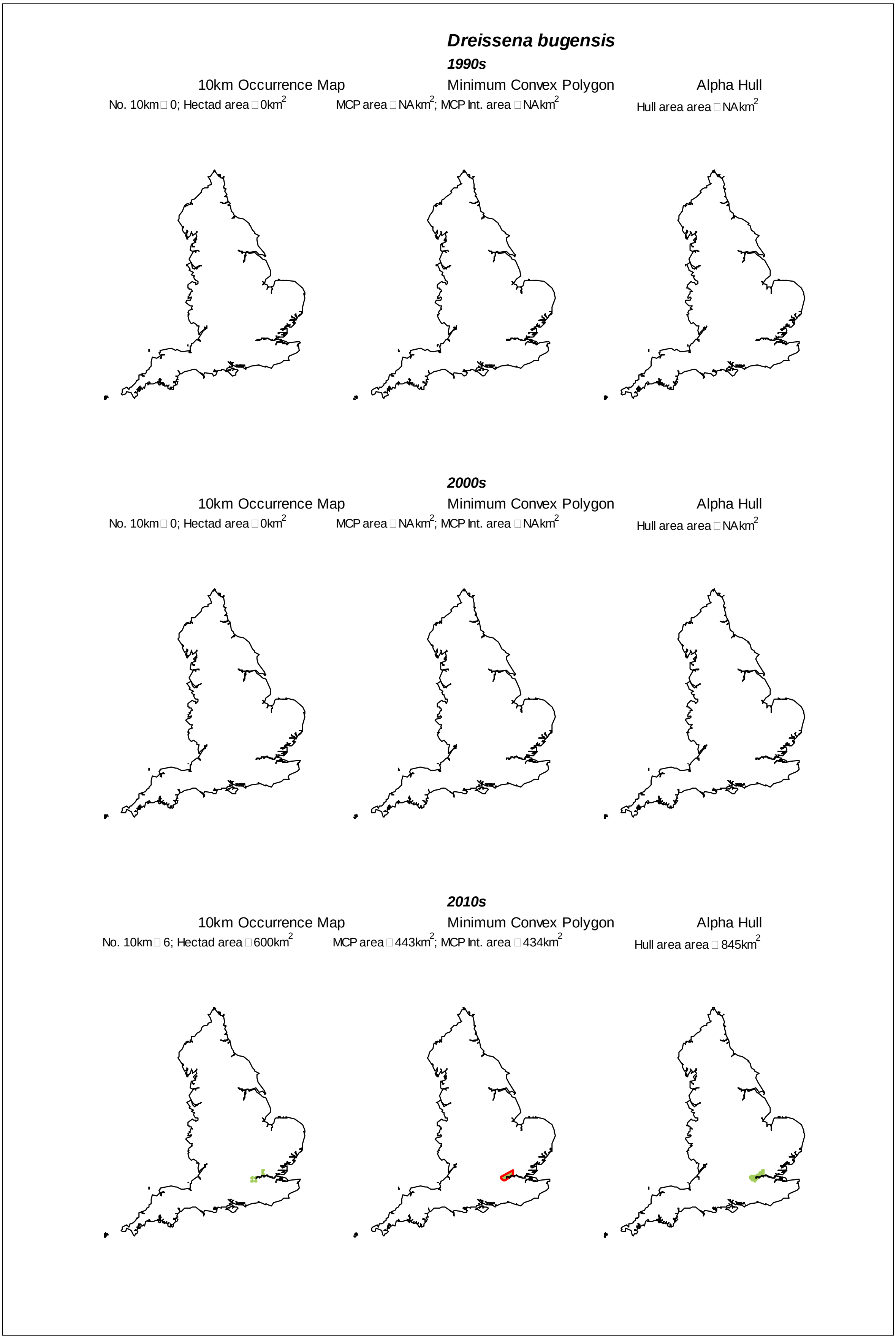


Figure C.20 Change in area of extent for *Dreissena bugensis* by decade (1990s to 2010s)

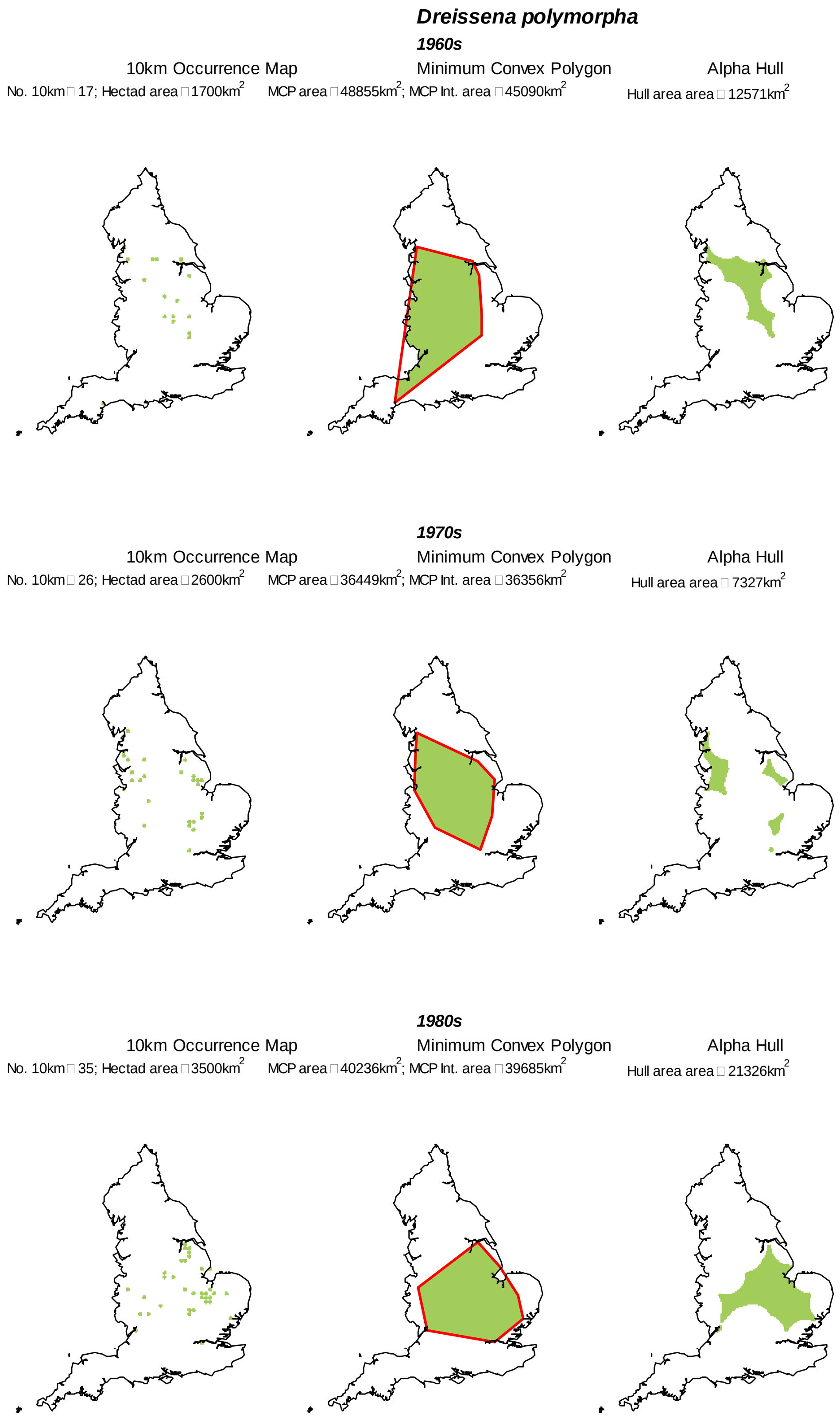


Figure C.21a Change in area of extent for *Dreissena polymorpha* by decade (1960s to 1980s)

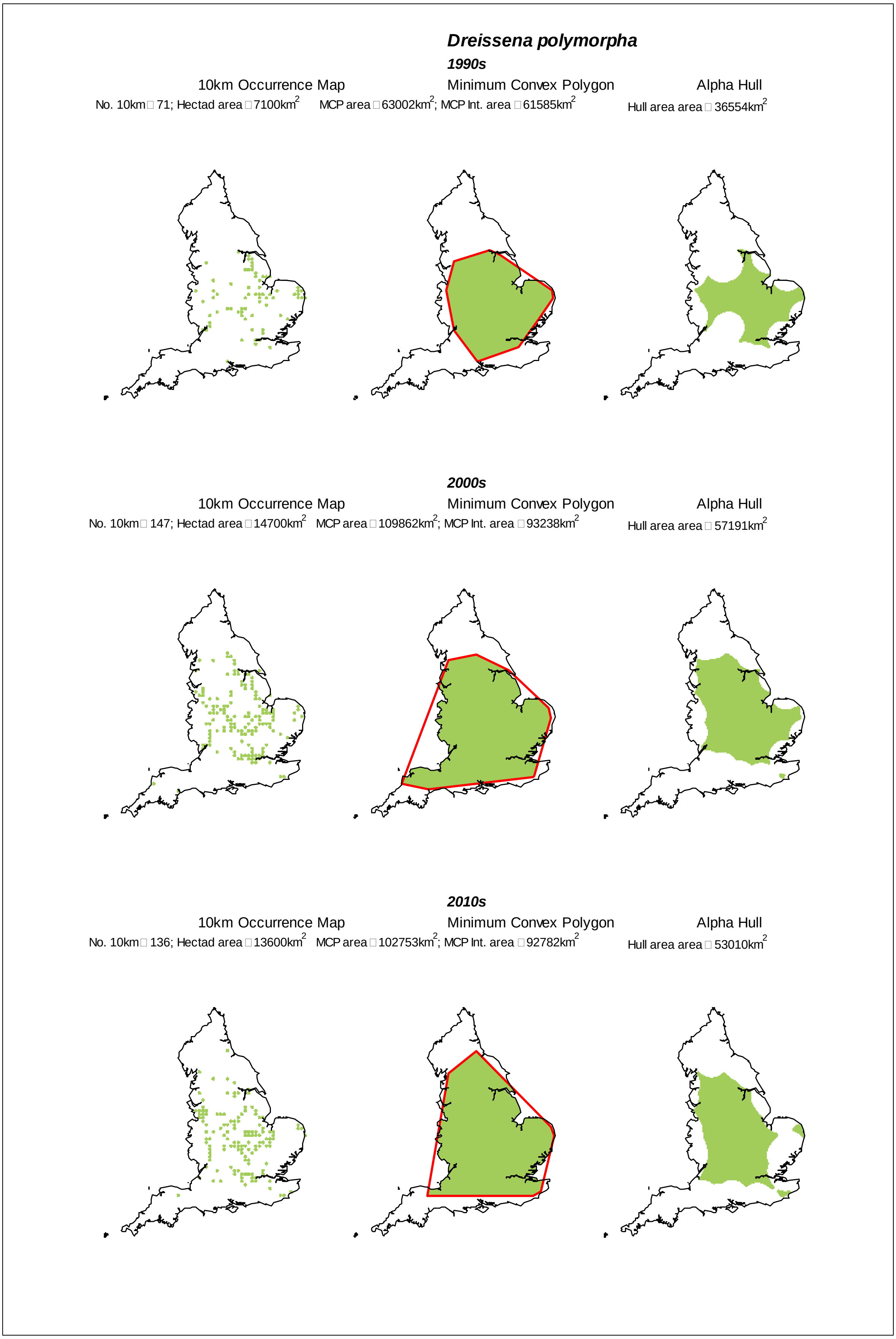


Figure C.21b Change in area of extent for *Dreissena polymorpha* by decade (1990s to 2010s)

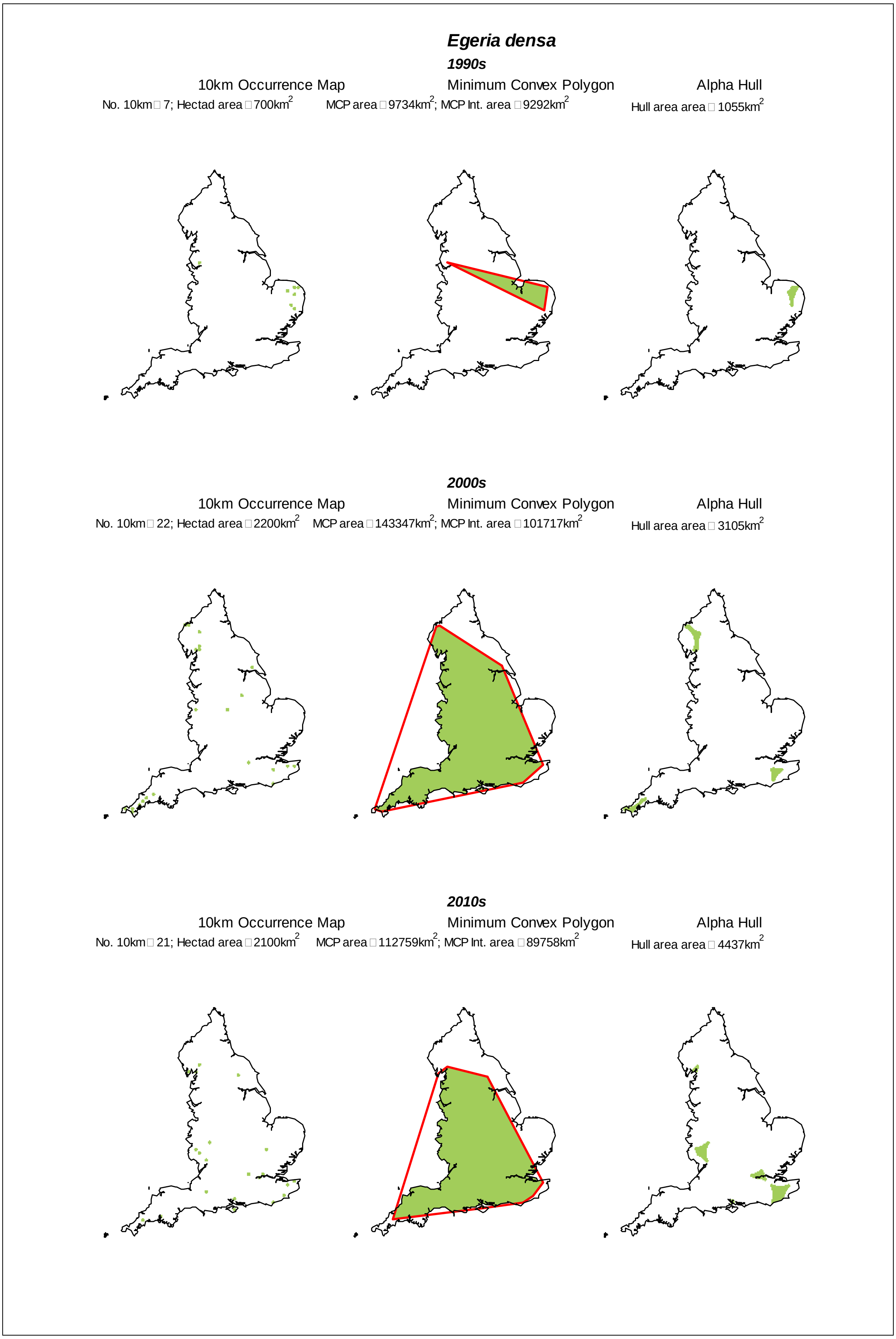


Figure C.22 Change in area of extent for *Egeria densa* by decade (1990s to 2010s)

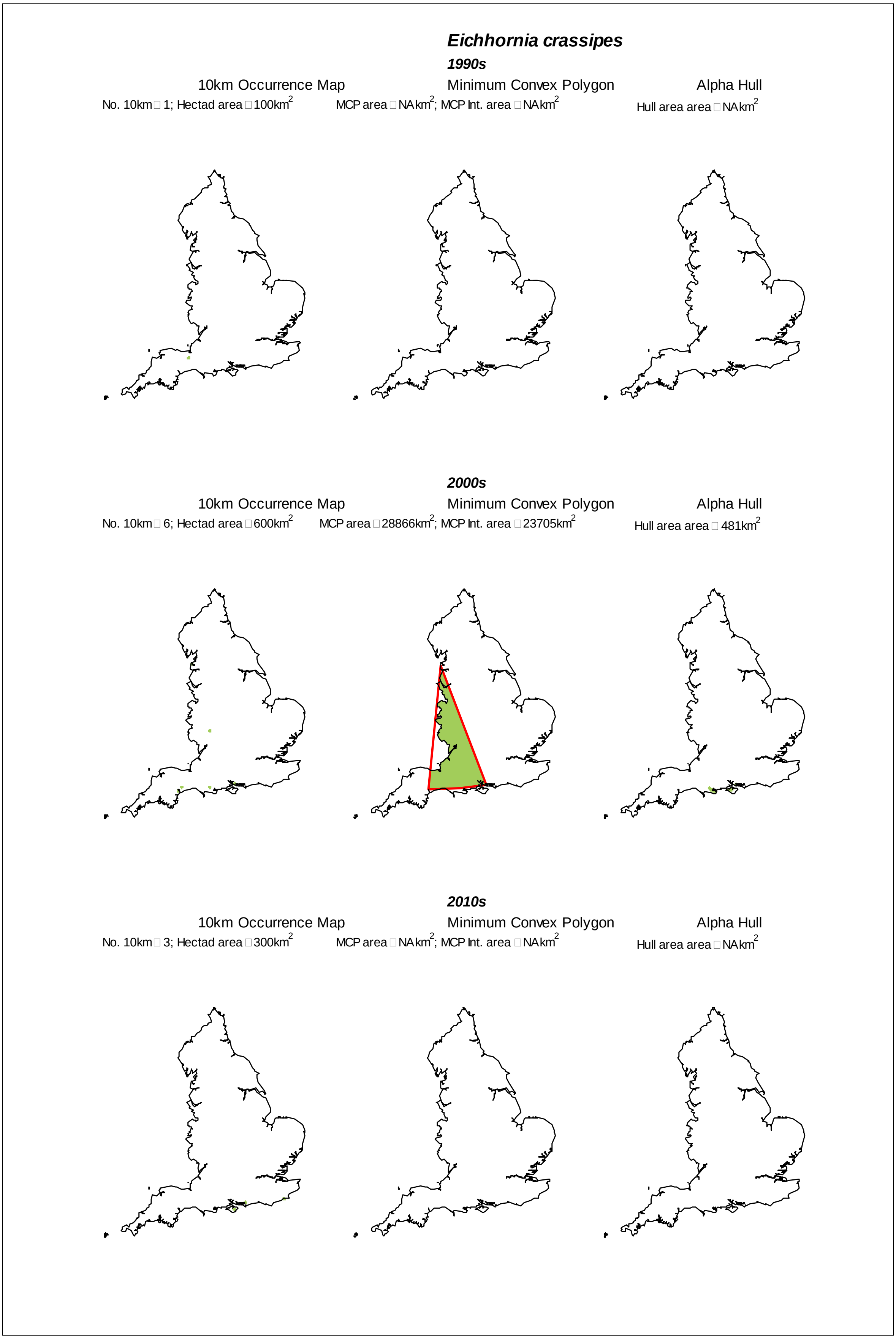


Figure C.23 Change in area of extent for *Eichhornia crassipes* by decade (1990s to 2010s)

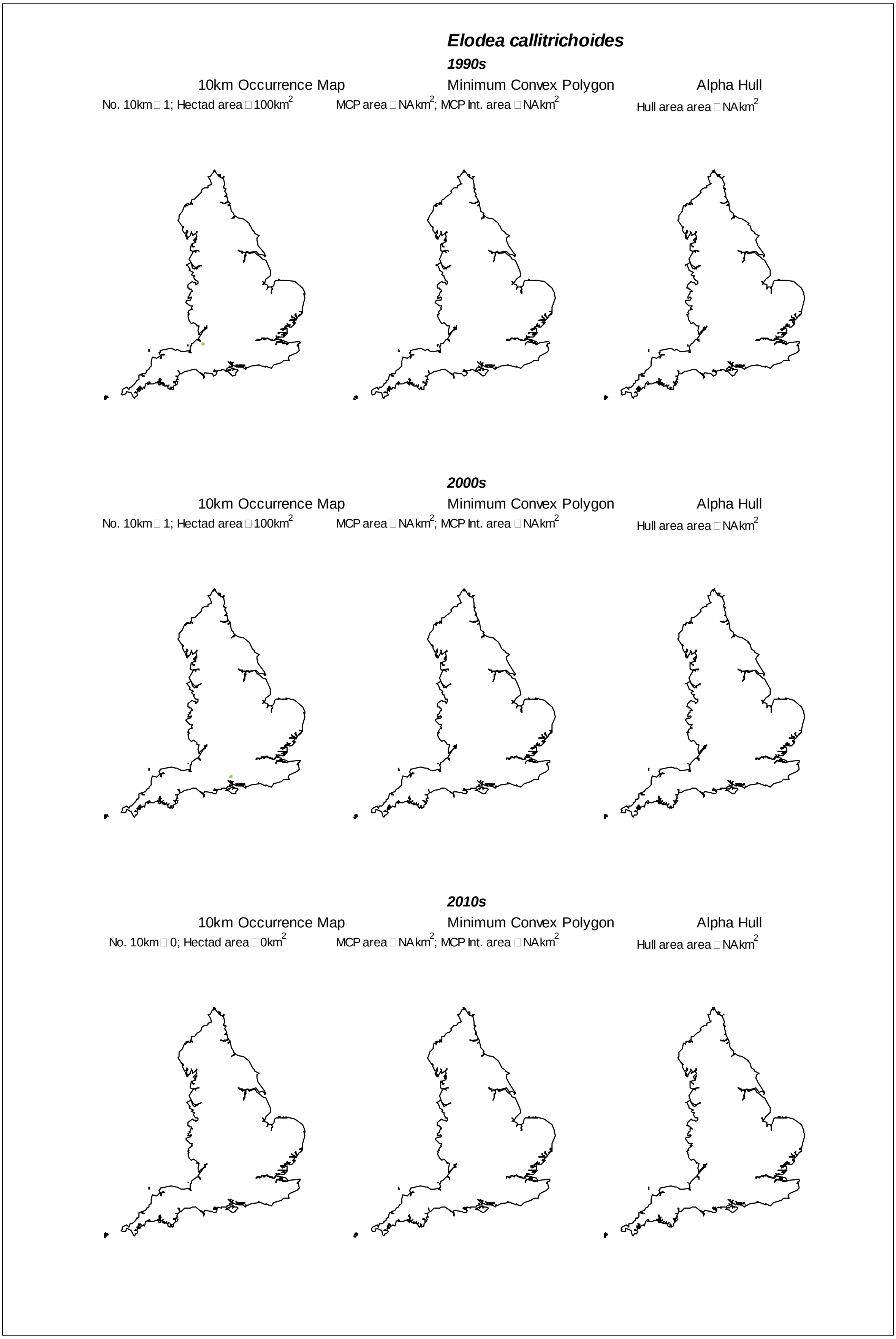


Figure C.24 Change in area of extent for *Elodea callitrichoides* by decade (1990s to 2010s)

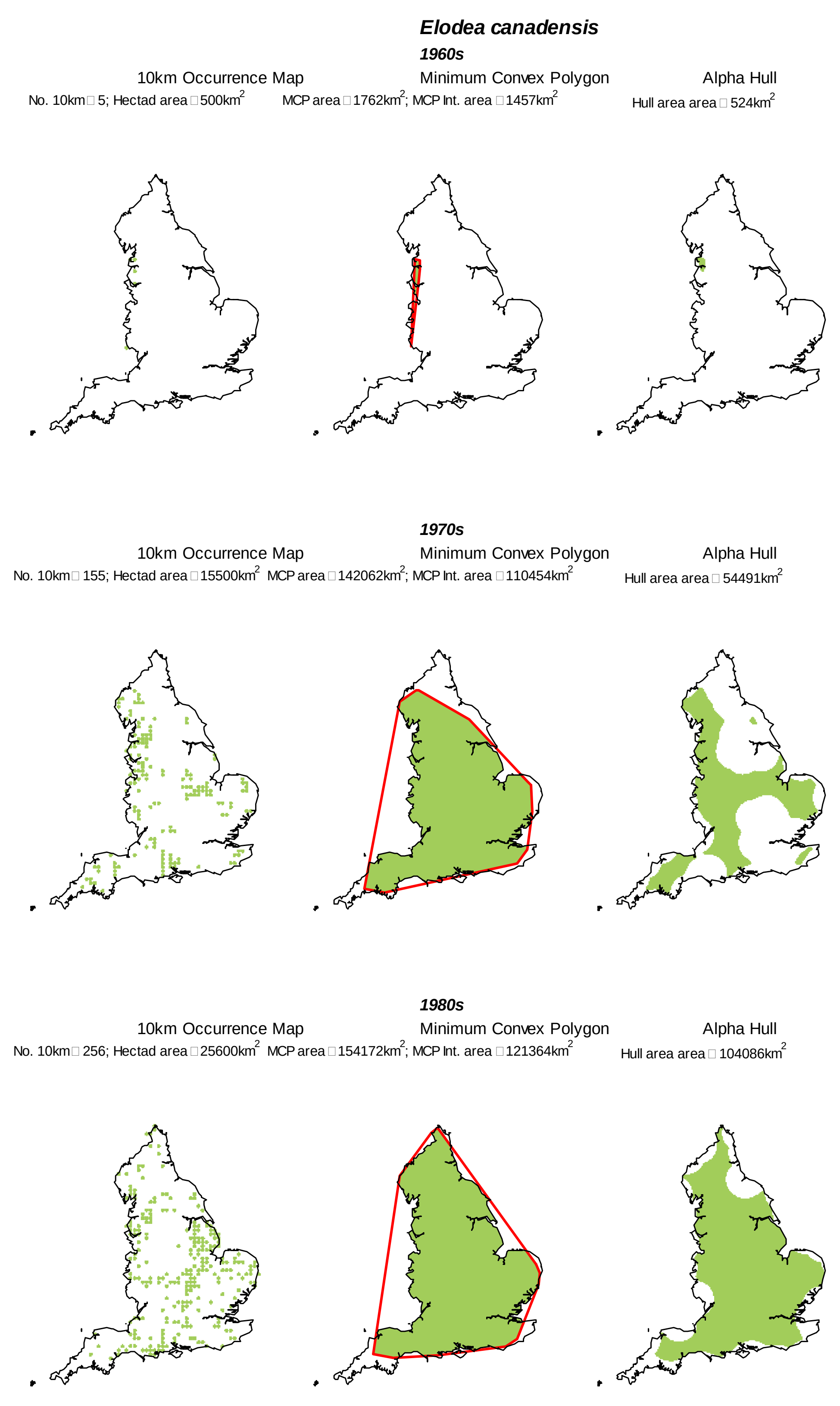


Figure C.25a Change in area of extent for *Elodea canadensis* by decade (1960s to 1980s)

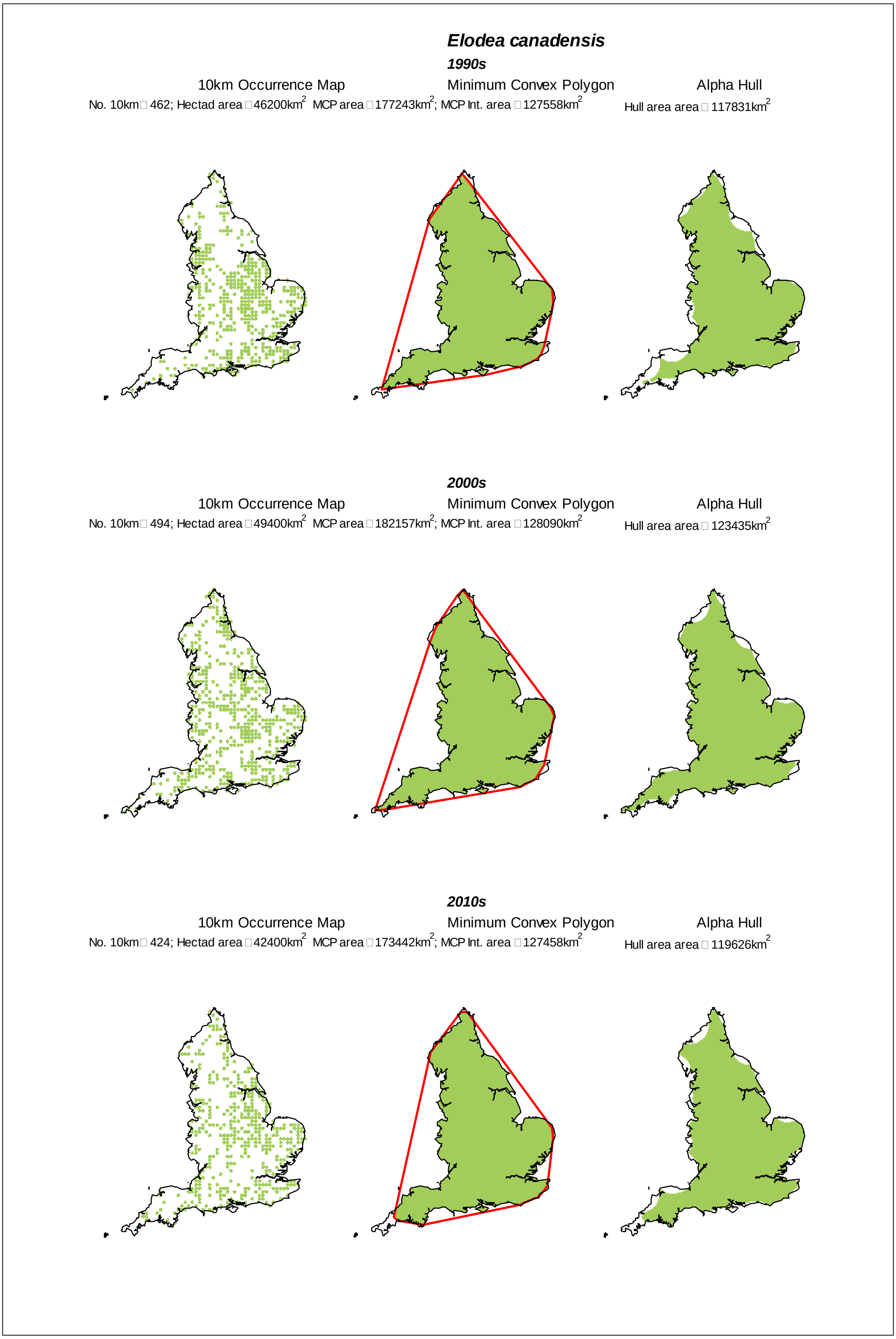


Figure C.25b Change in area of extent for *Elodea canadensis* by decade (1990s to 2010s)

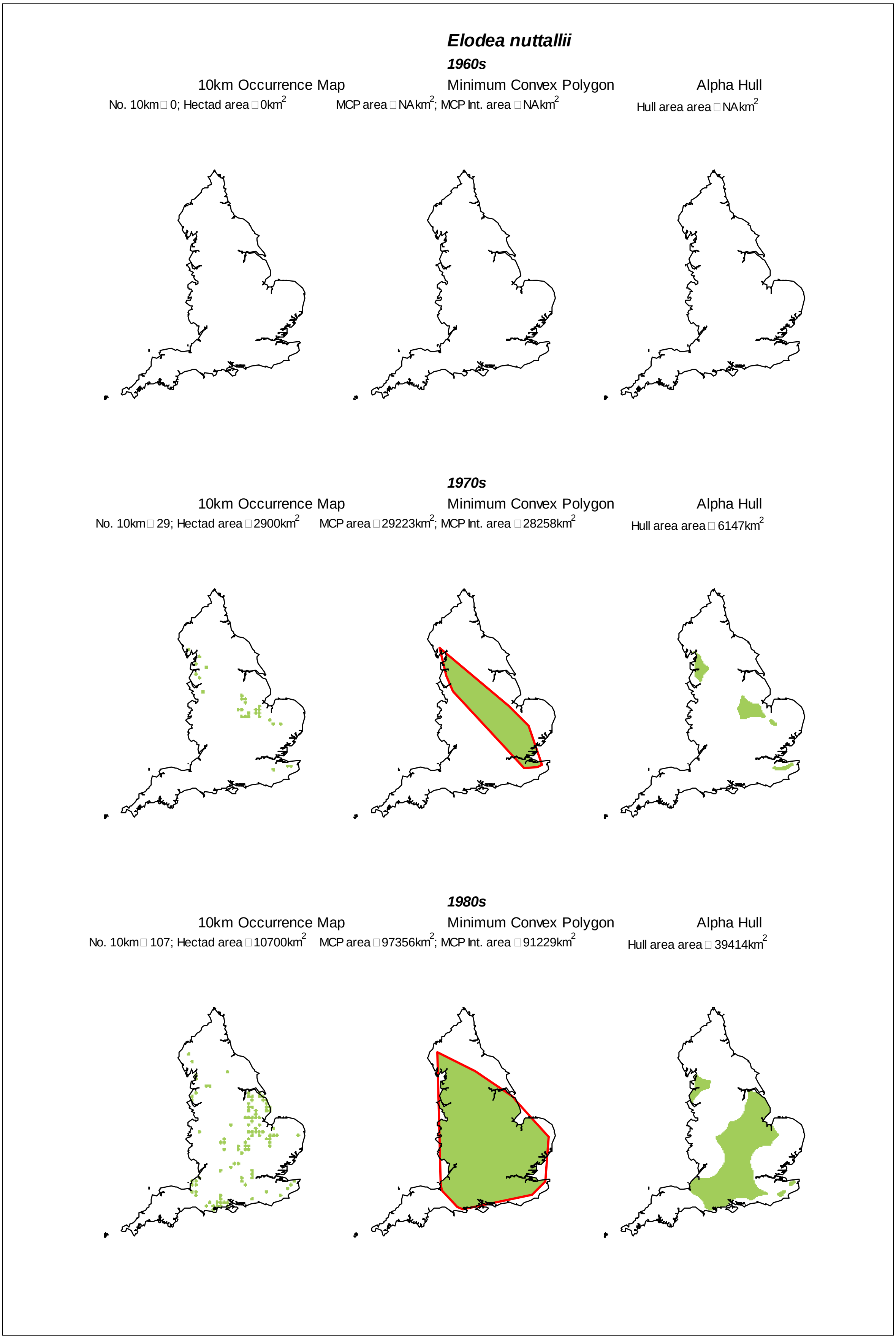


Figure C.26a Change in area of extent for *Elodea nuttallii* by decade (1960s to 1980s)

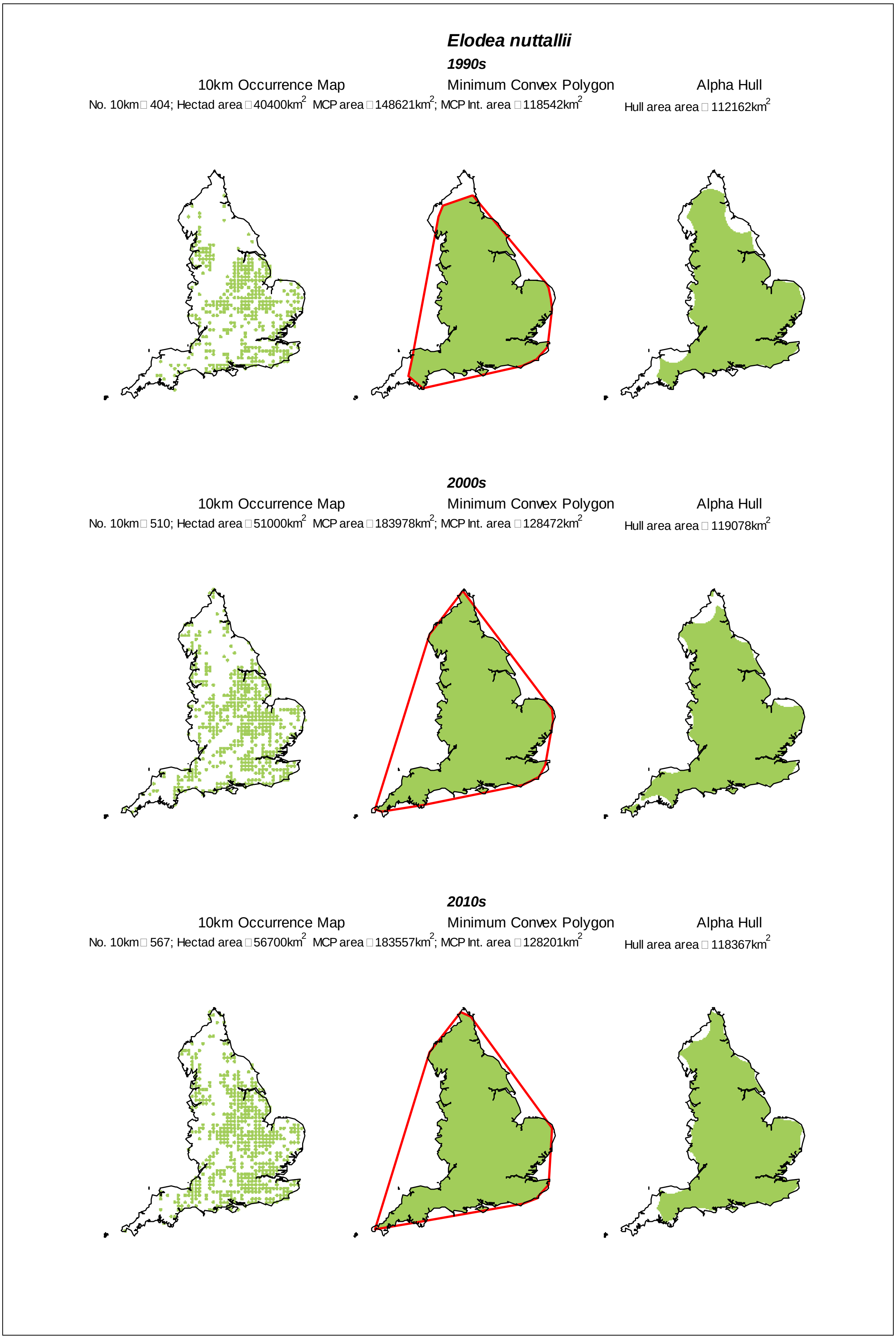


Figure C.26b Change in area of extent for *Elodea nuttallii* by decade (1990s to 2010s)

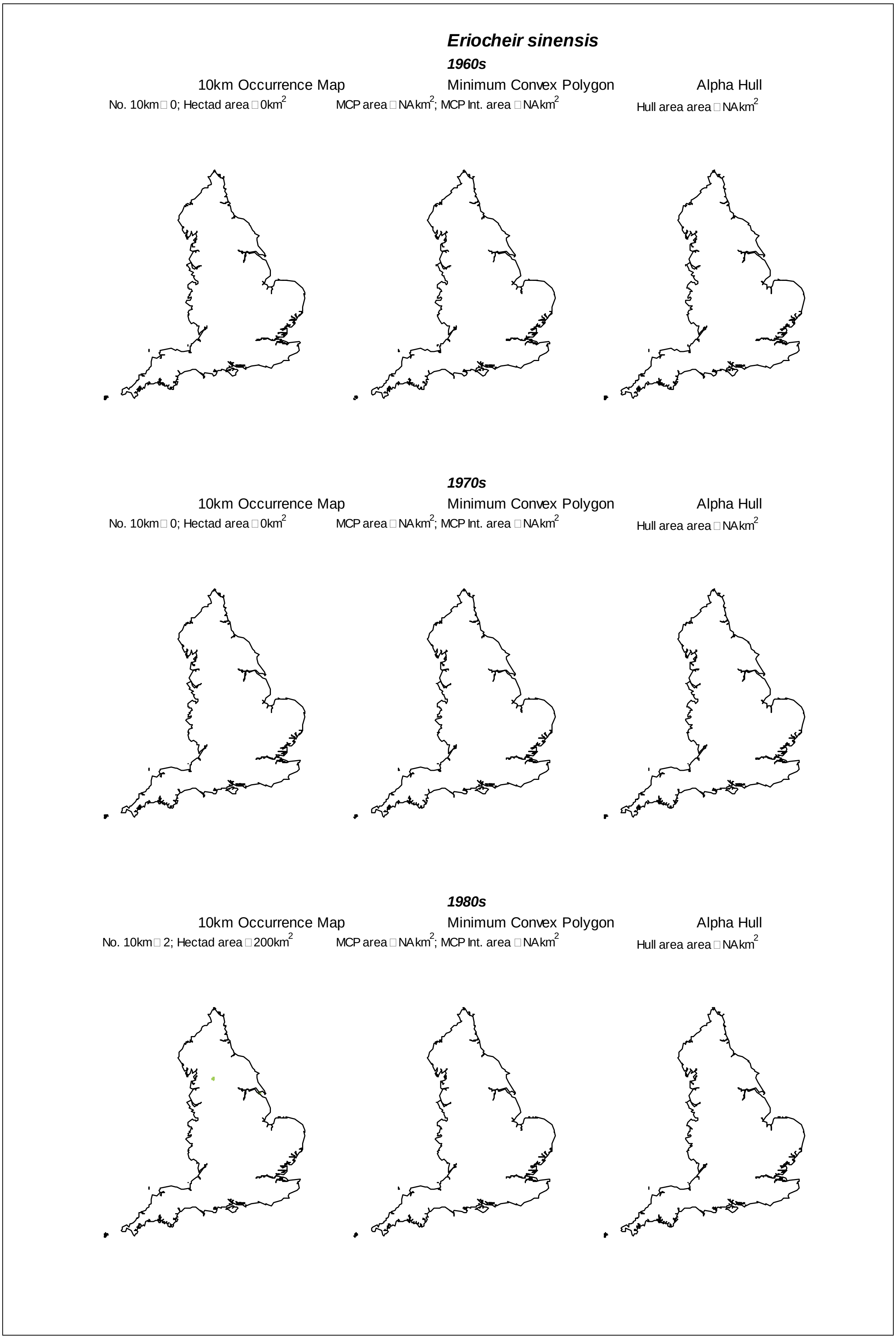


Figure C.27a Change in area of extent for *Eriocheir sinensis* by decade (1960s to 1980s)

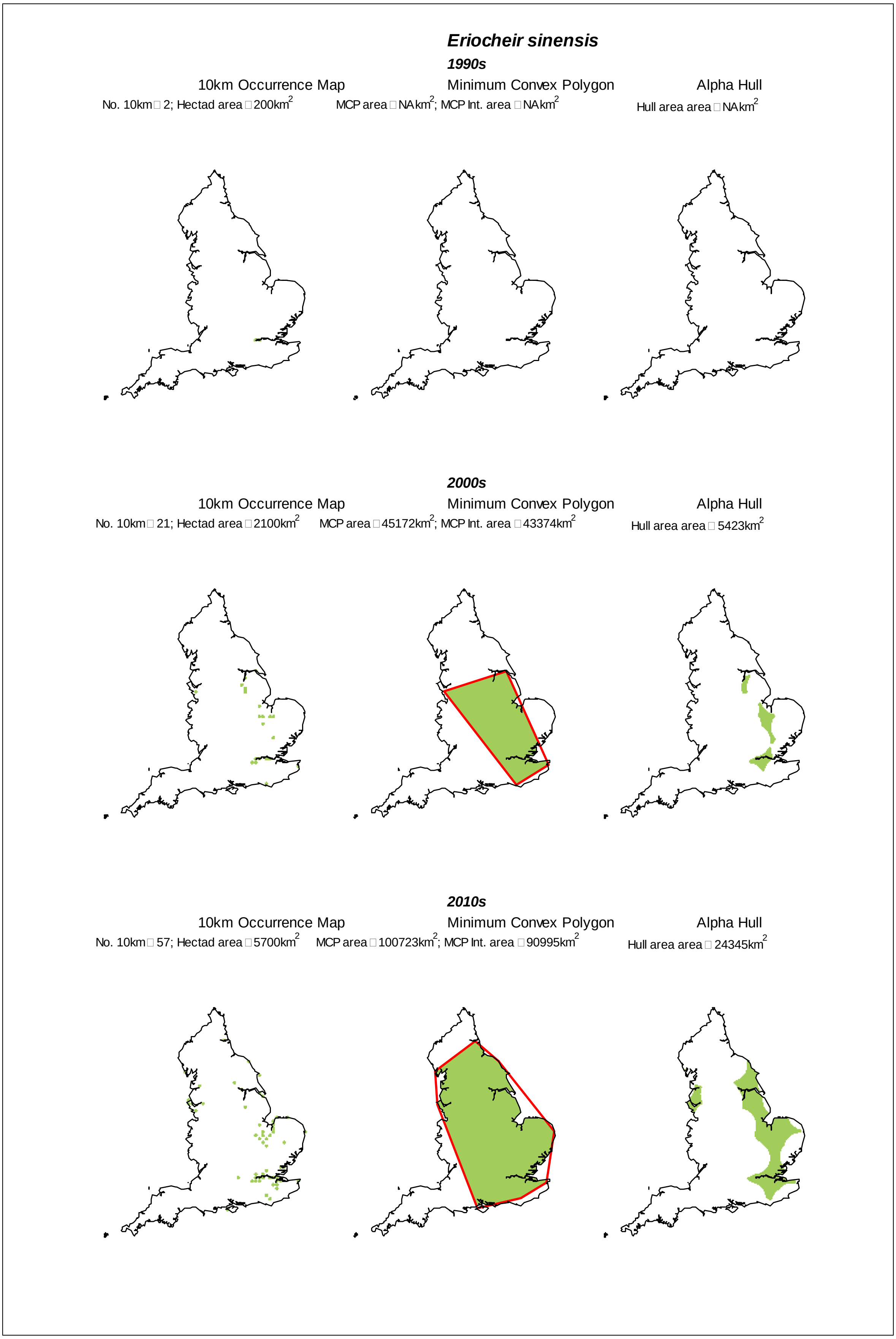


Figure C.27b Change in area of extent for *Eriocheir sinensis* by decade (1990s to 2010s)

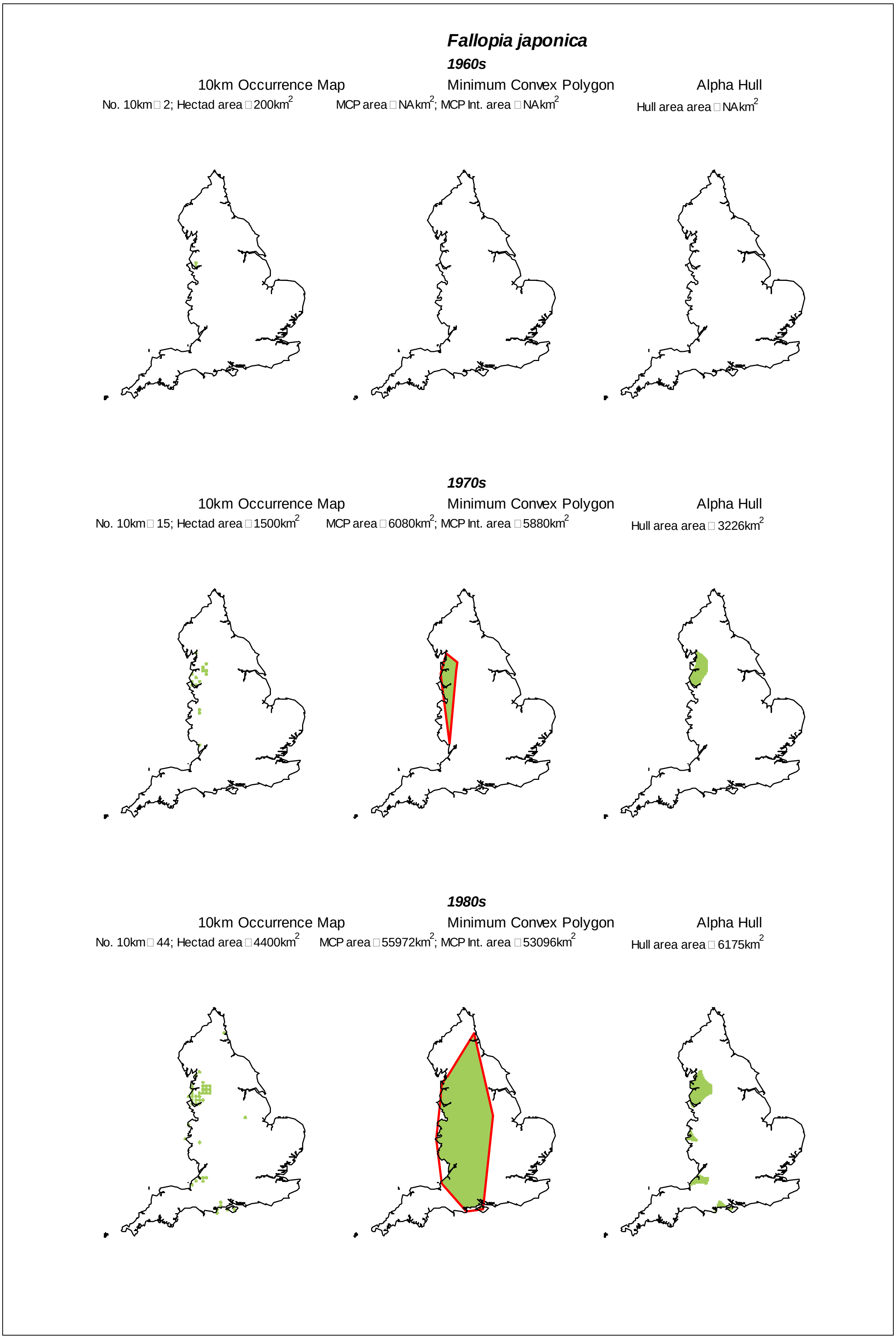


Figure C.28a Change in area of extent for *Fallopia japonica* by decade (1960s to 1980s)

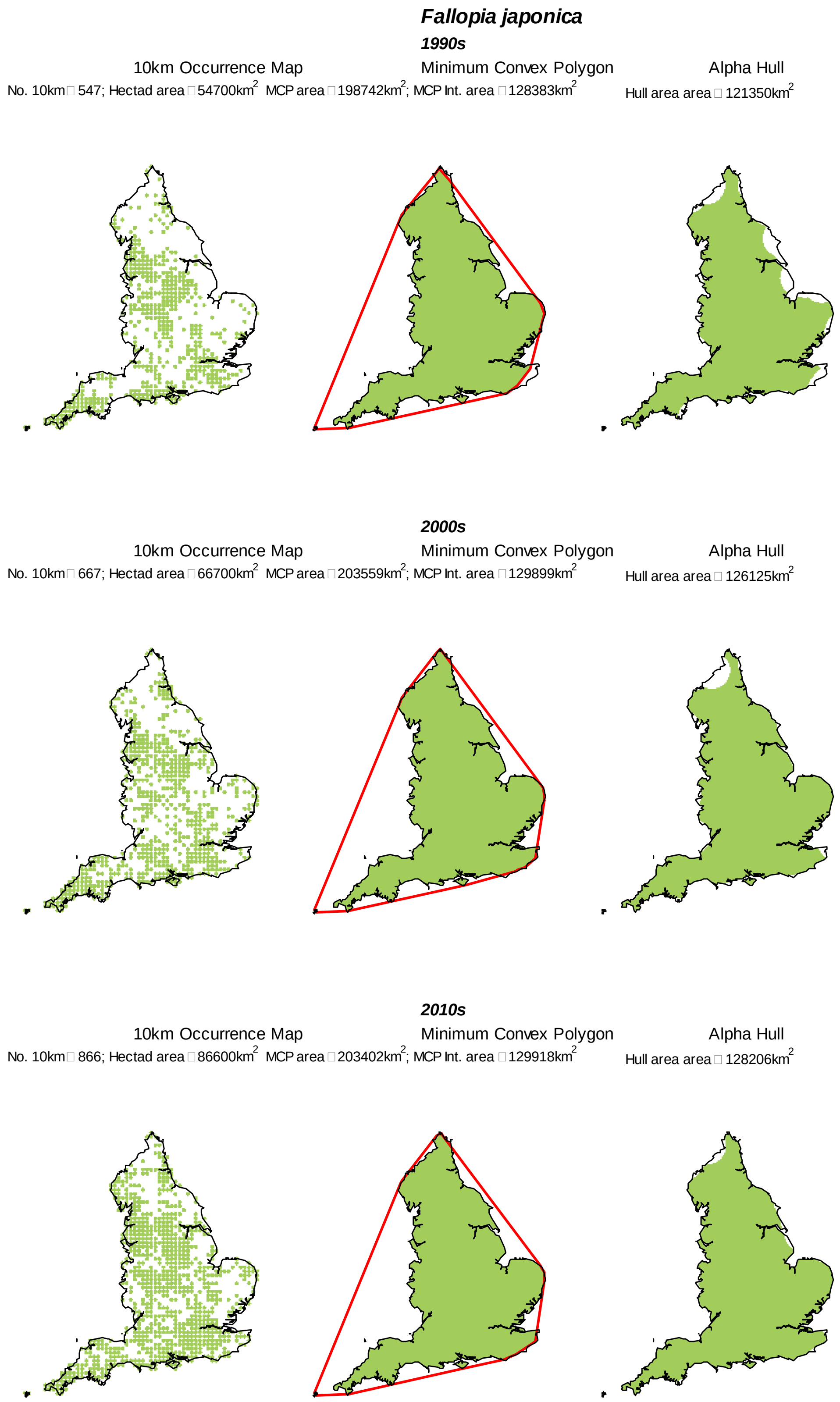
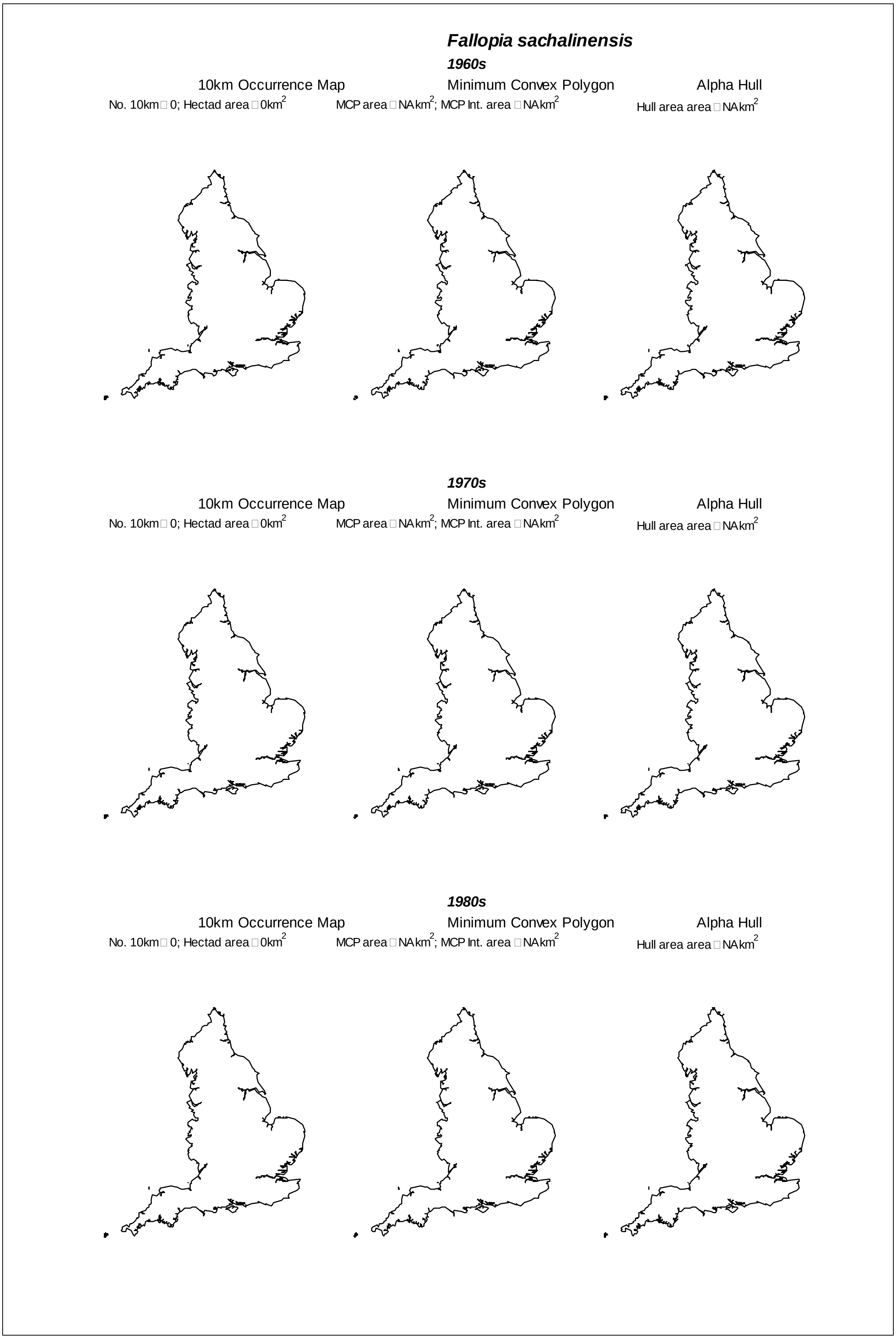


Figure C.28b Change in area of extent for *Fallopia japonica* by decade (1990s to 2010s)



Fallopia sachalinensis

1960s

10km Occurrence Map — No. 10km□ 0; Hectad area□0km^2

Minimum Convex Polygon — MCP area□NAkm^2; MCP Int. area □NAkm^2

Alpha Hull — Hull area area□NAkm^2

1970s

10km Occurrence Map — No. 10km□ 0; Hectad area□0km^2

Minimum Convex Polygon — MCP area□NAkm^2; MCP Int. area □NAkm^2

Alpha Hull — Hull area area□NAkm^2

1980s

10km Occurrence Map — No. 10km□ 0; Hectad area□0km^2

Minimum Convex Polygon — MCP area□NAkm^2; MCP Int. area □NAkm^2

Alpha Hull — Hull area area□NAkm^2

Figure C.29a Change in area of extent for *Fallopia sachalinensis* by decade (1960s to 1980s)

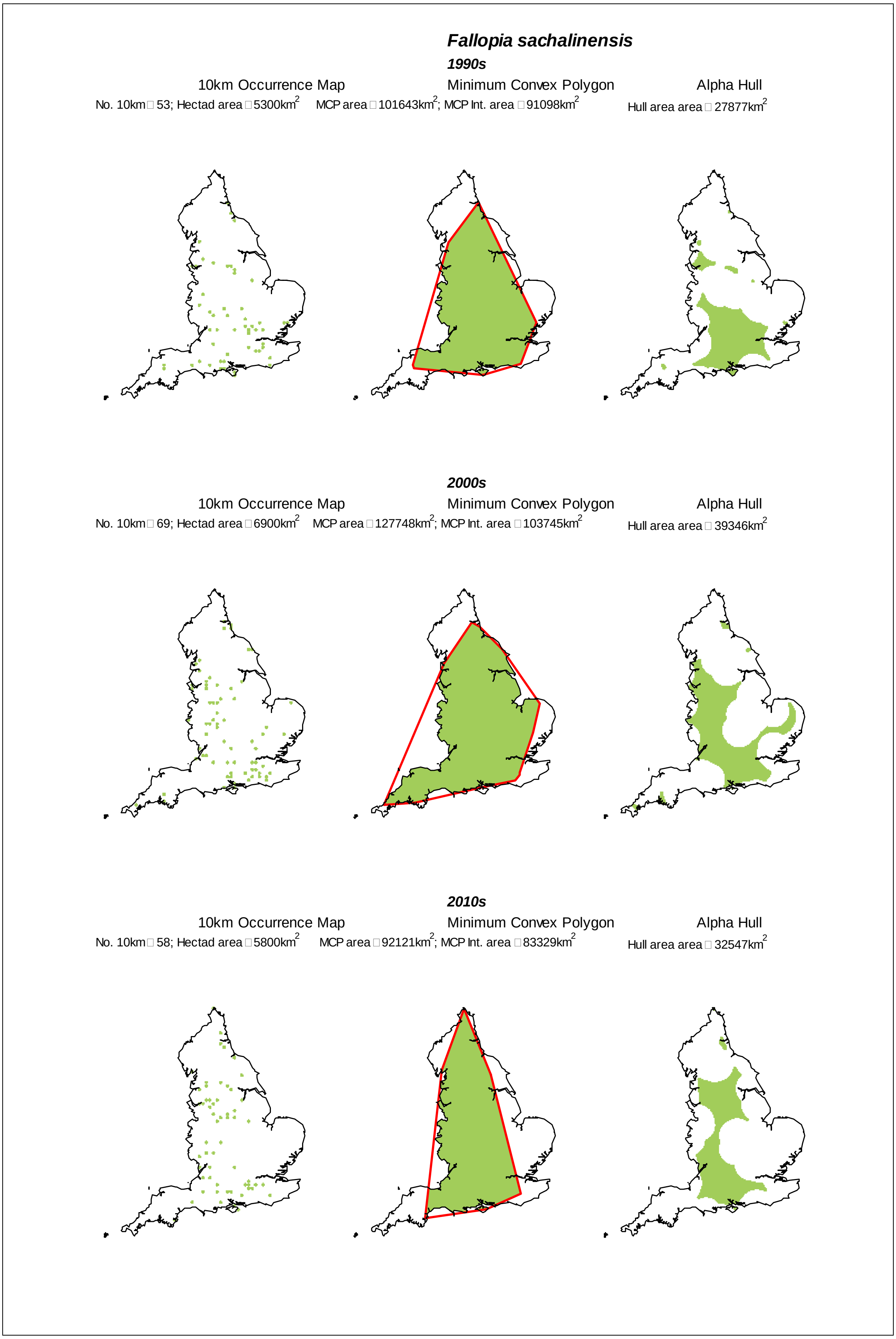


Figure C.29b Change in area of extent for *Fallopia sachalinensis* by decade (1990s to 2010s)

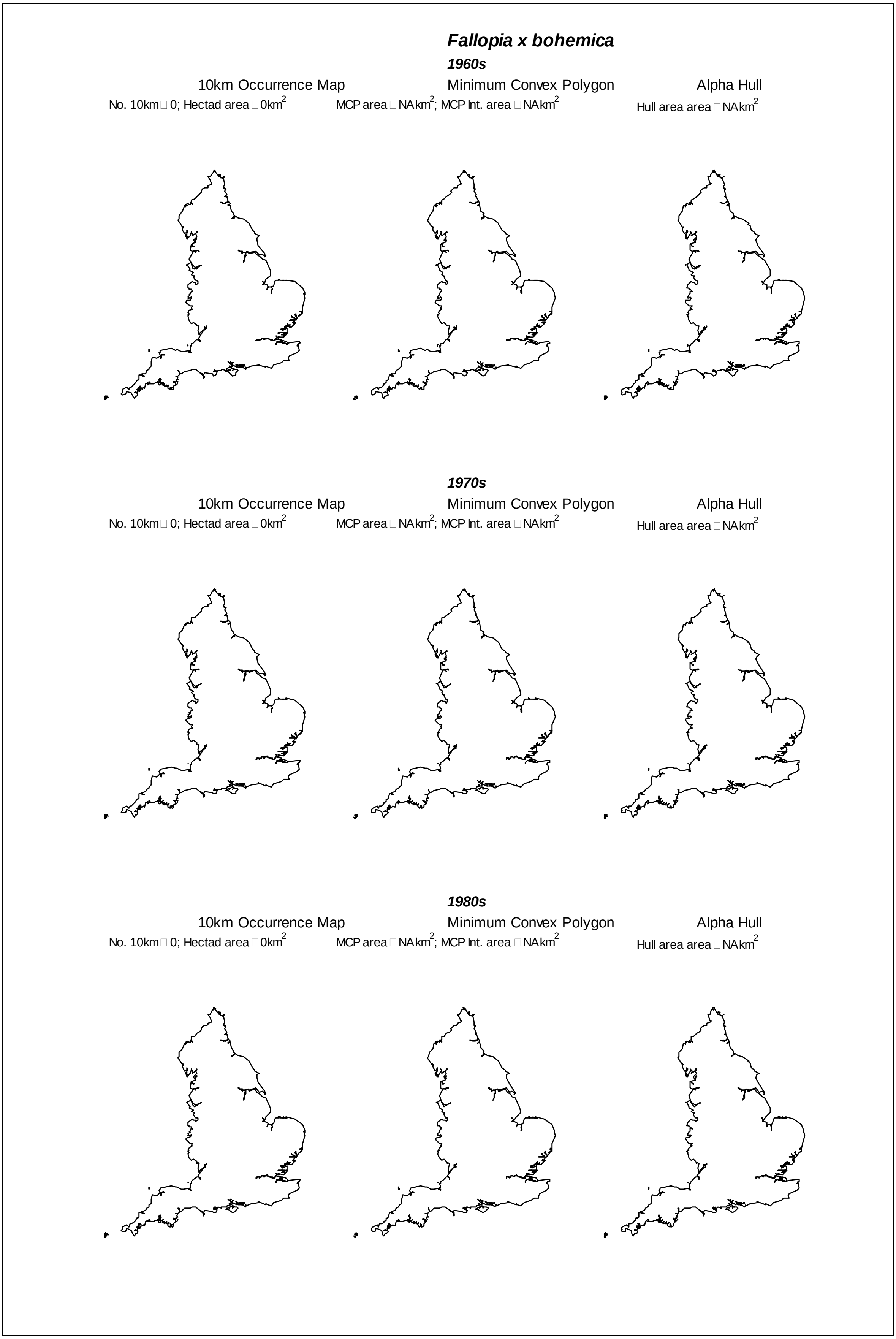


Figure C.30a Change in area of extent for *Fallopia x bohemica* by decade (1960s to 1980s)

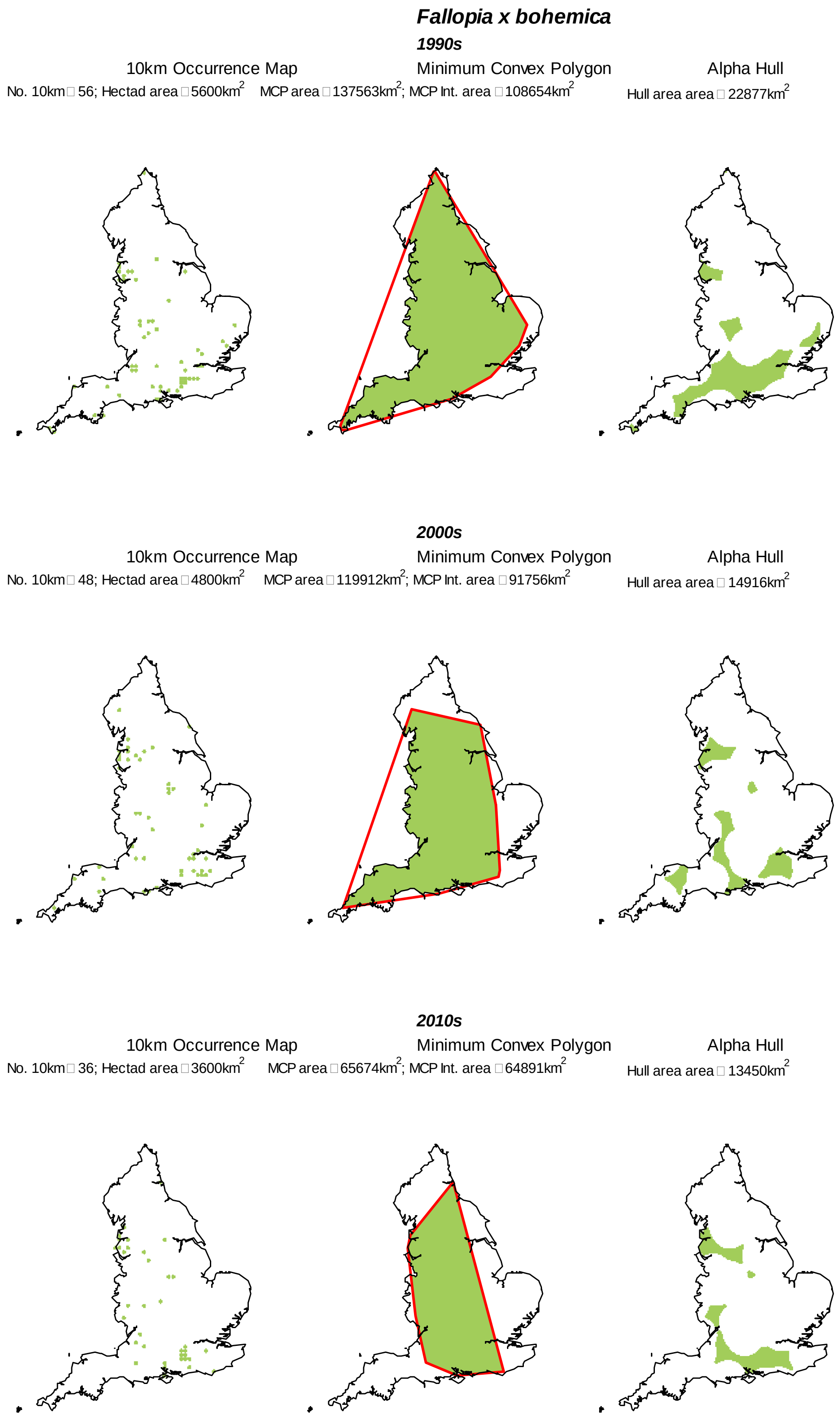


Figure C.30b Change in area of extent for *Fallopia x bohemica* by decade (1990s to 2010s)

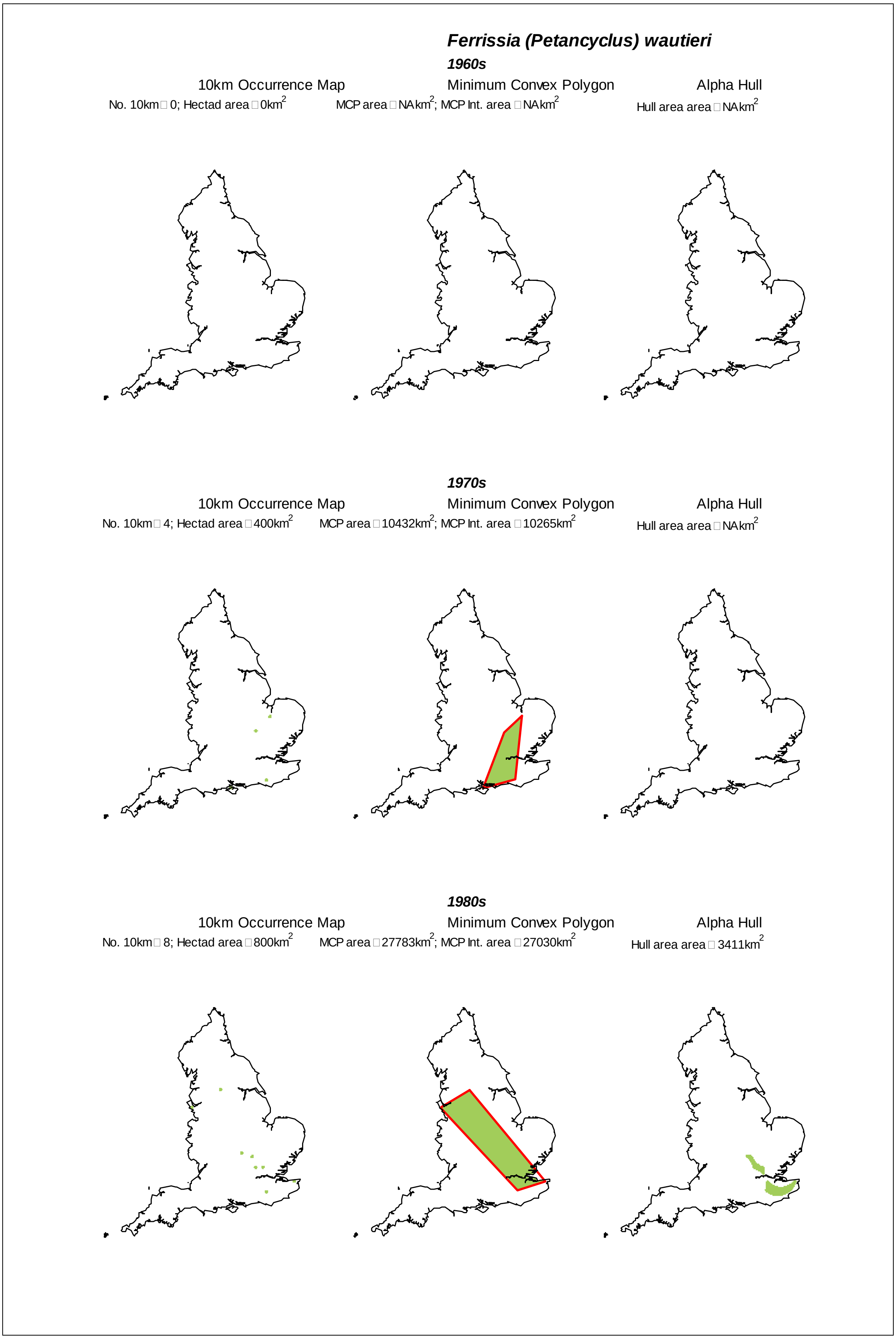


Figure C.31a Change in area of extent for *Ferrissia (Petancyclus) wautieri* by decade (1960s to 1980s)

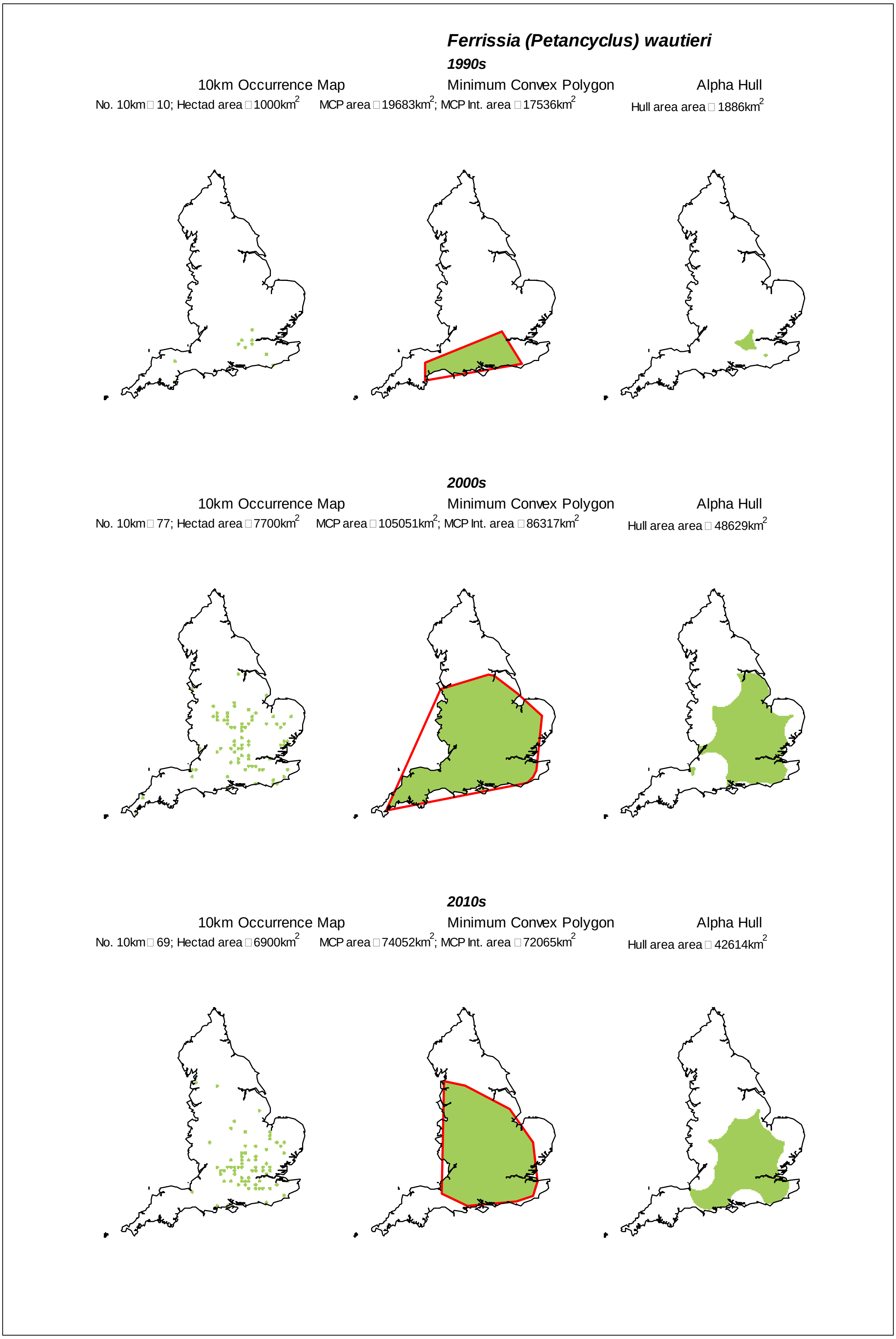


Figure C.31b Change in area of extent for *Ferrissia (Petancyclus) wautieri* by decade (1990s to 2010s)

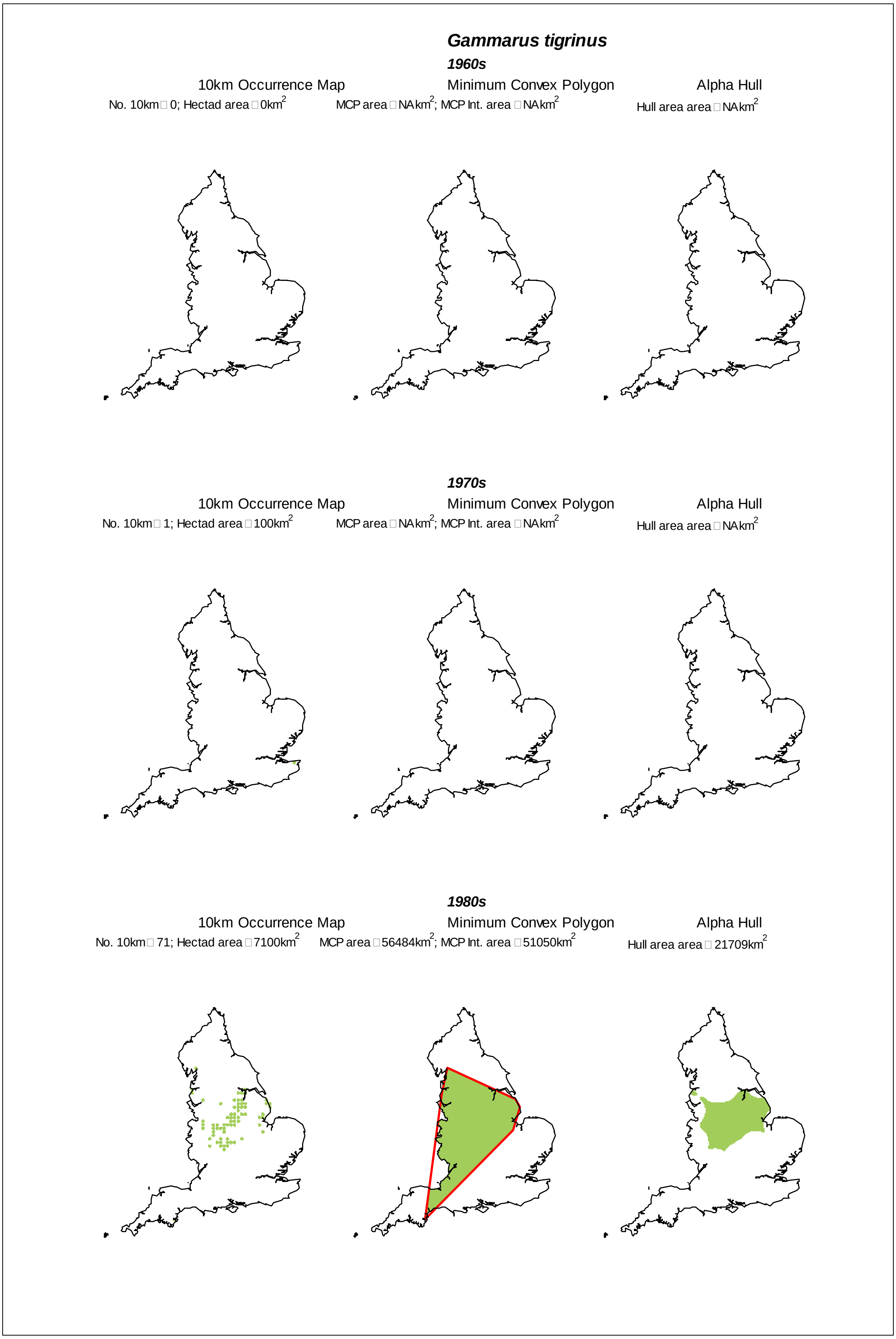
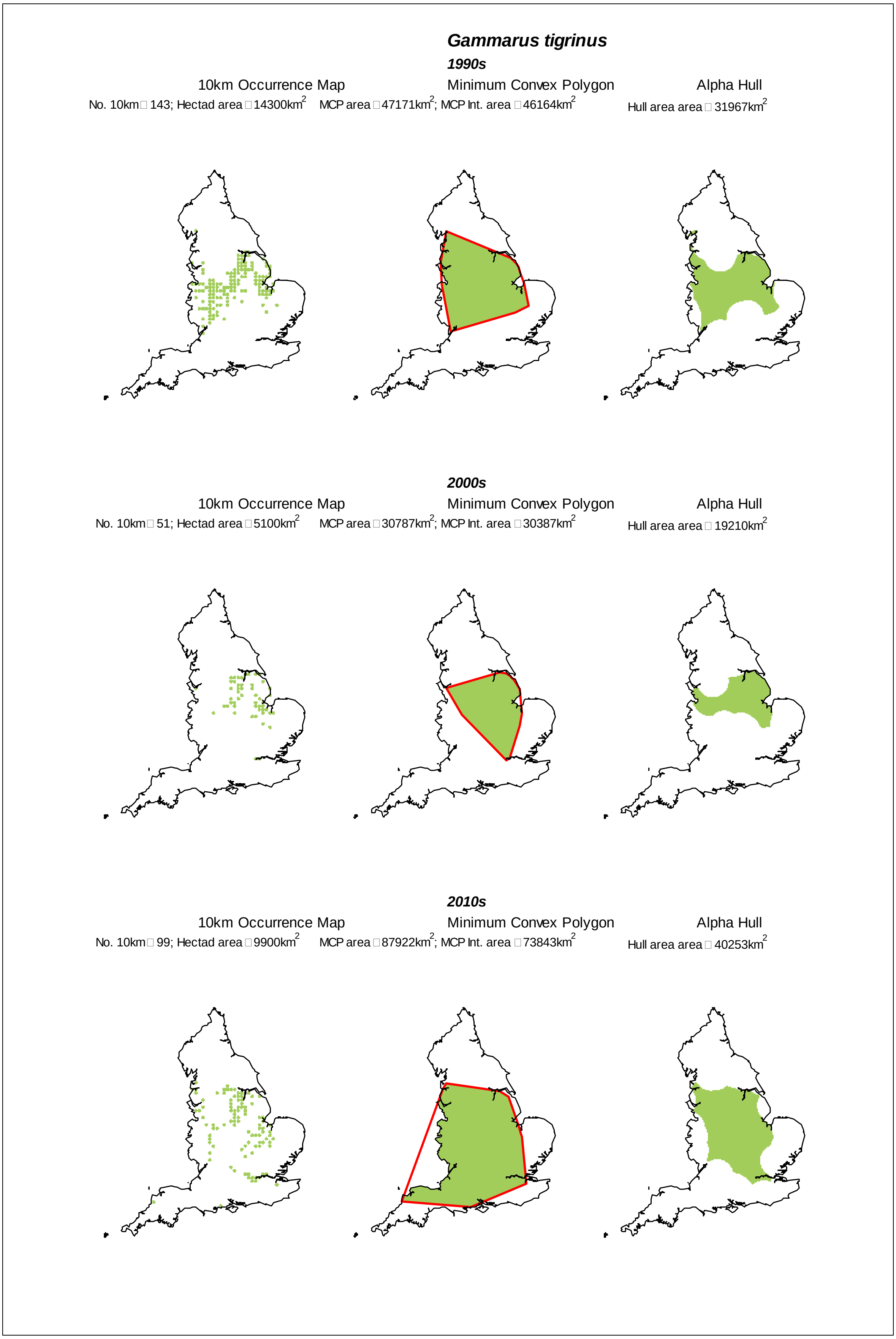


Figure C.32a Change in area of extent for *Gammarus tigrinus* by decade (1960s to 1980s)



Gammarus tigrinus

1990s

10km Occurrence Map
No. 10km□ 143; Hectad area □14300km^2

Minimum Convex Polygon
MCP area □47171km^2; MCP Int. area □46164km^2

Alpha Hull
Hull area area □ 31967km^2

2000s

10km Occurrence Map
No. 10km□ 51; Hectad area □5100km^2

Minimum Convex Polygon
MCP area □30787km^2; MCP Int. area □30387km^2

Alpha Hull
Hull area area □ 19210km^2

2010s

10km Occurrence Map
No. 10km□ 99; Hectad area □9900km^2

Minimum Convex Polygon
MCP area □87922km^2; MCP Int. area □73843km^2

Alpha Hull
Hull area area □ 40253km^2

Figure C.32b Change in area of extent for *Gammarus tigrinus* by decade (1990s to 2010s)

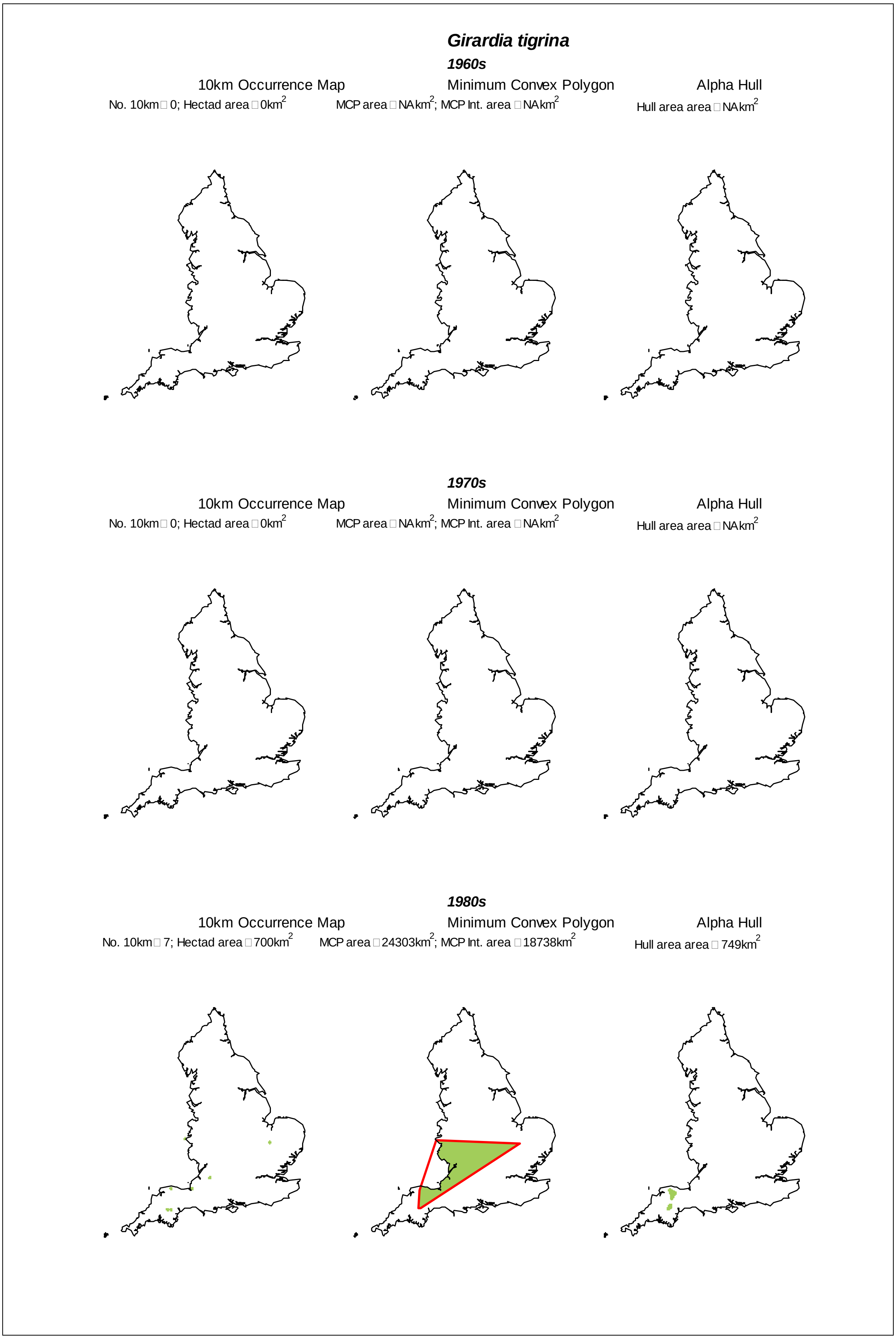


Figure C.33a Change in area of extent for *Girardia tigrina* by decade (1960s to 1980s)

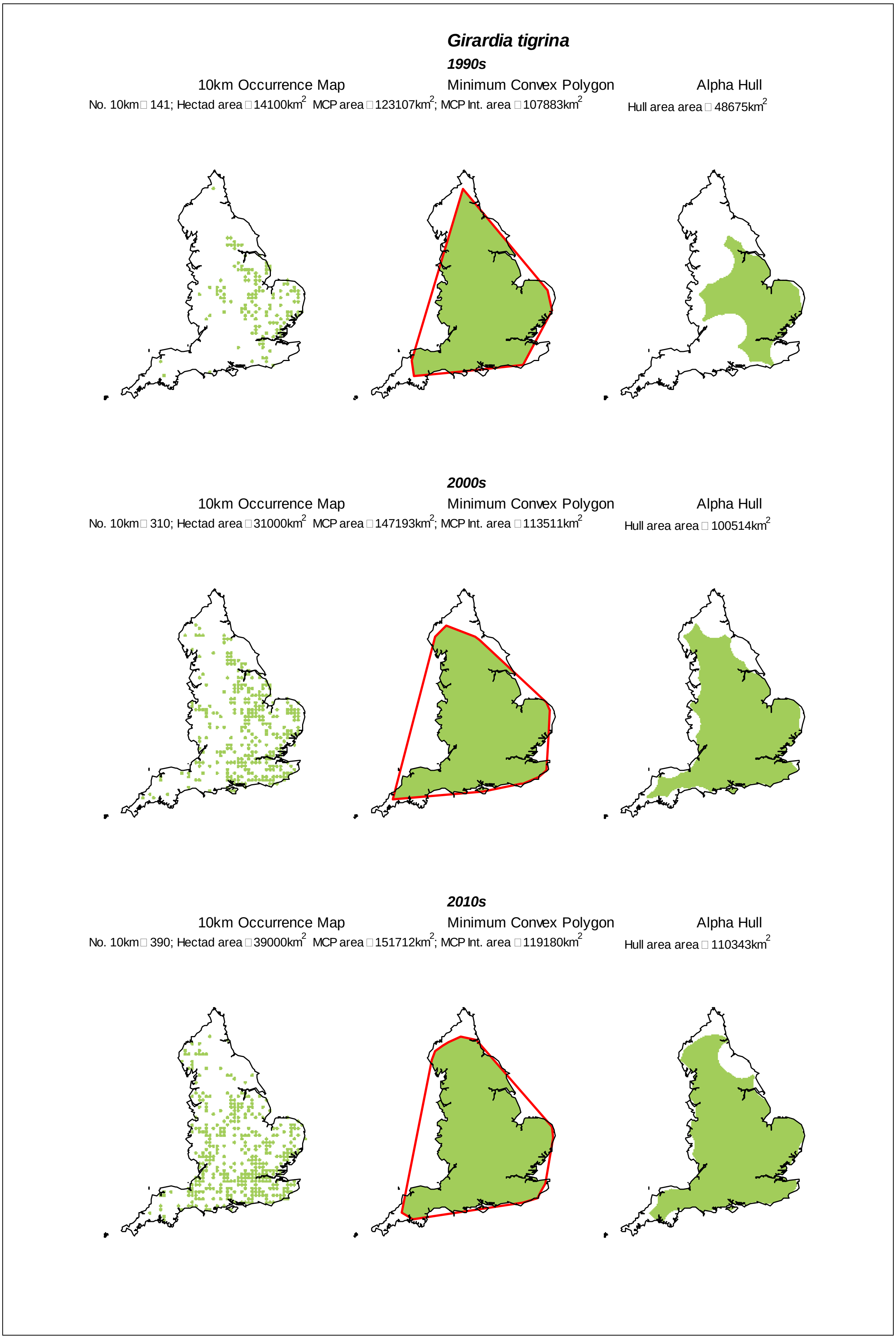


Figure C.33b Change in area of extent for *Girardia tigrina* by decade (1990s to 2010s)

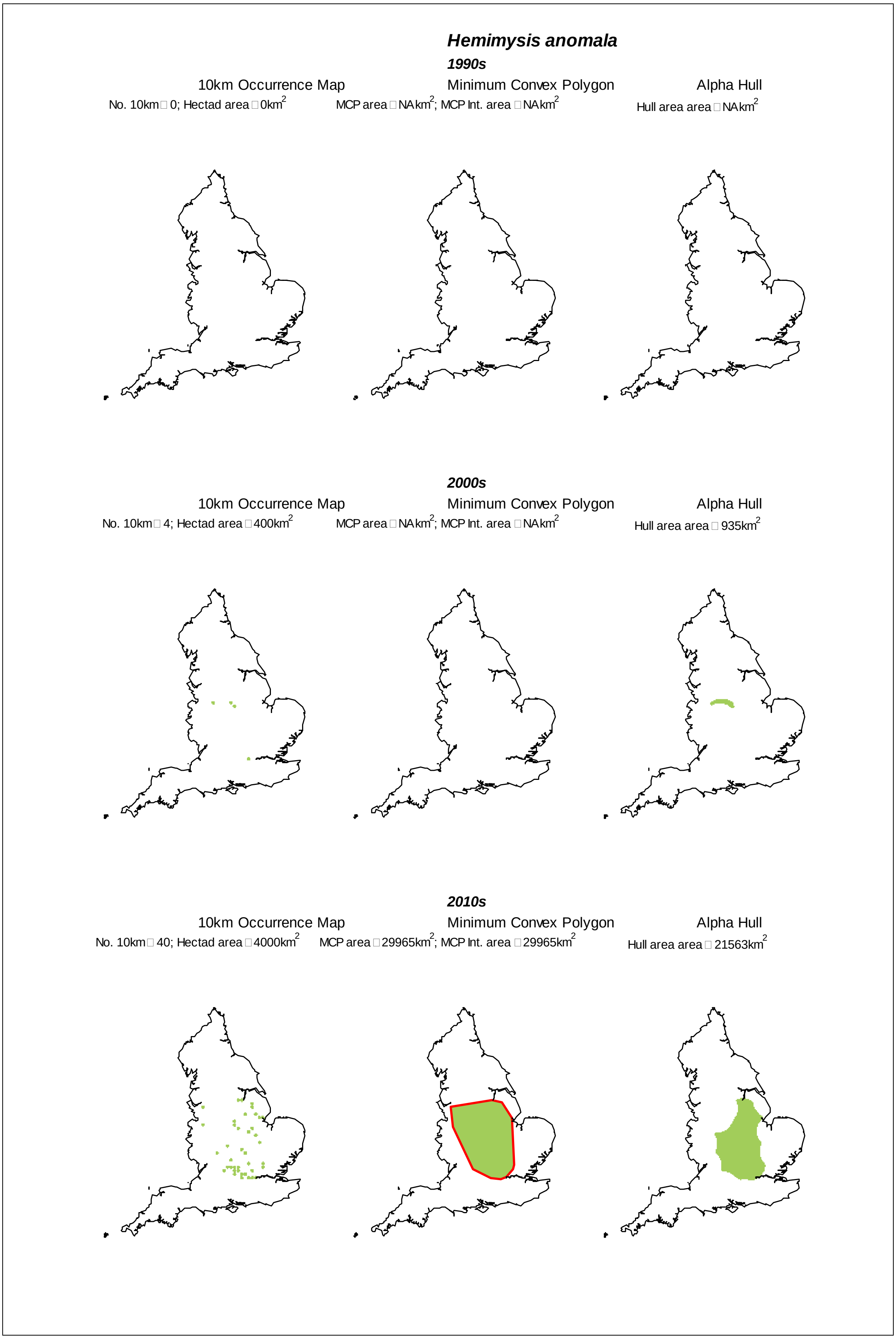


Figure C.34 Change in area of extent for *Hemimysis anomala* by decade (1990s to 2010s)

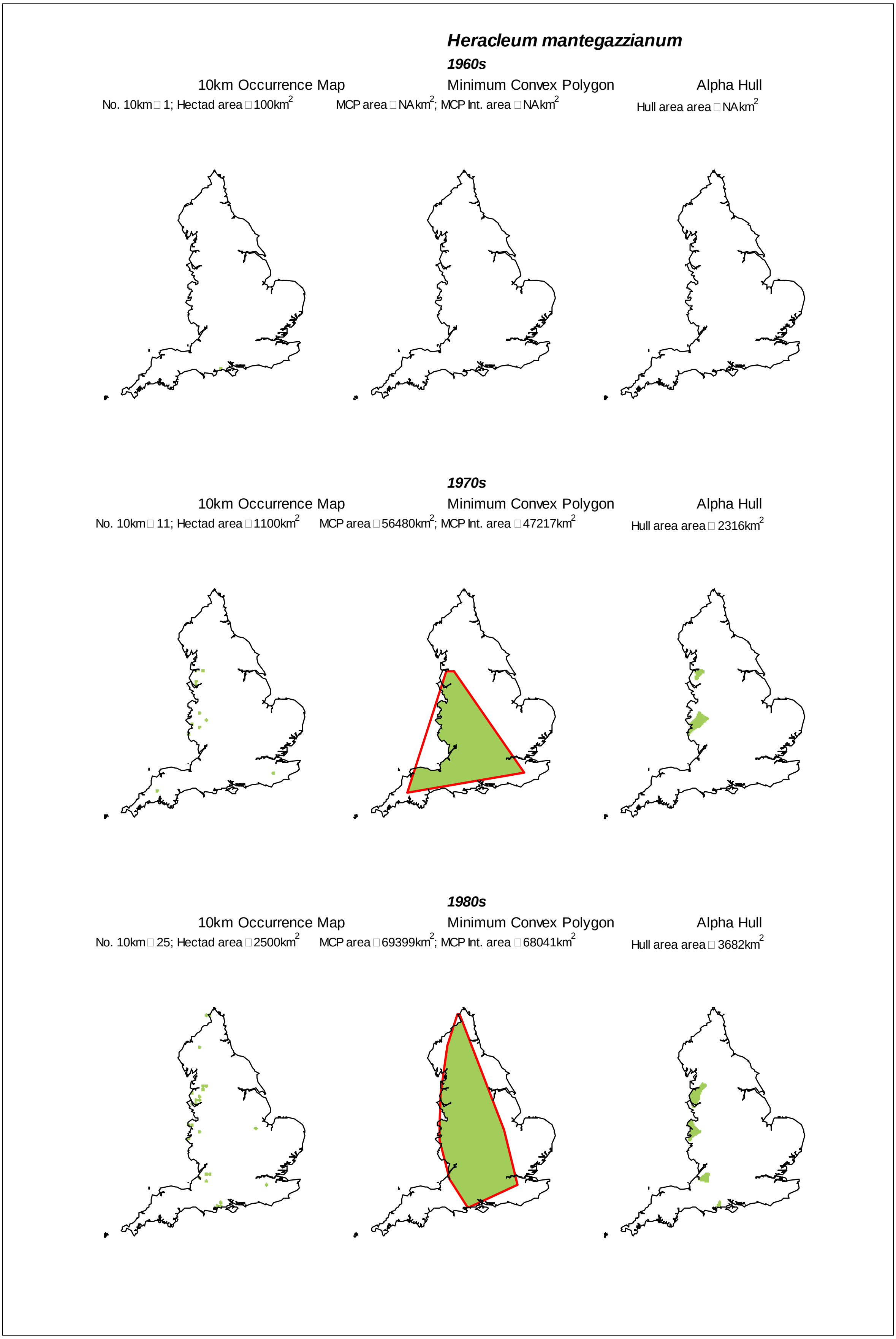


Figure C.35a Change in area of extent for *Heracleum mantegazzianum* by decade (1960s to 1980s)

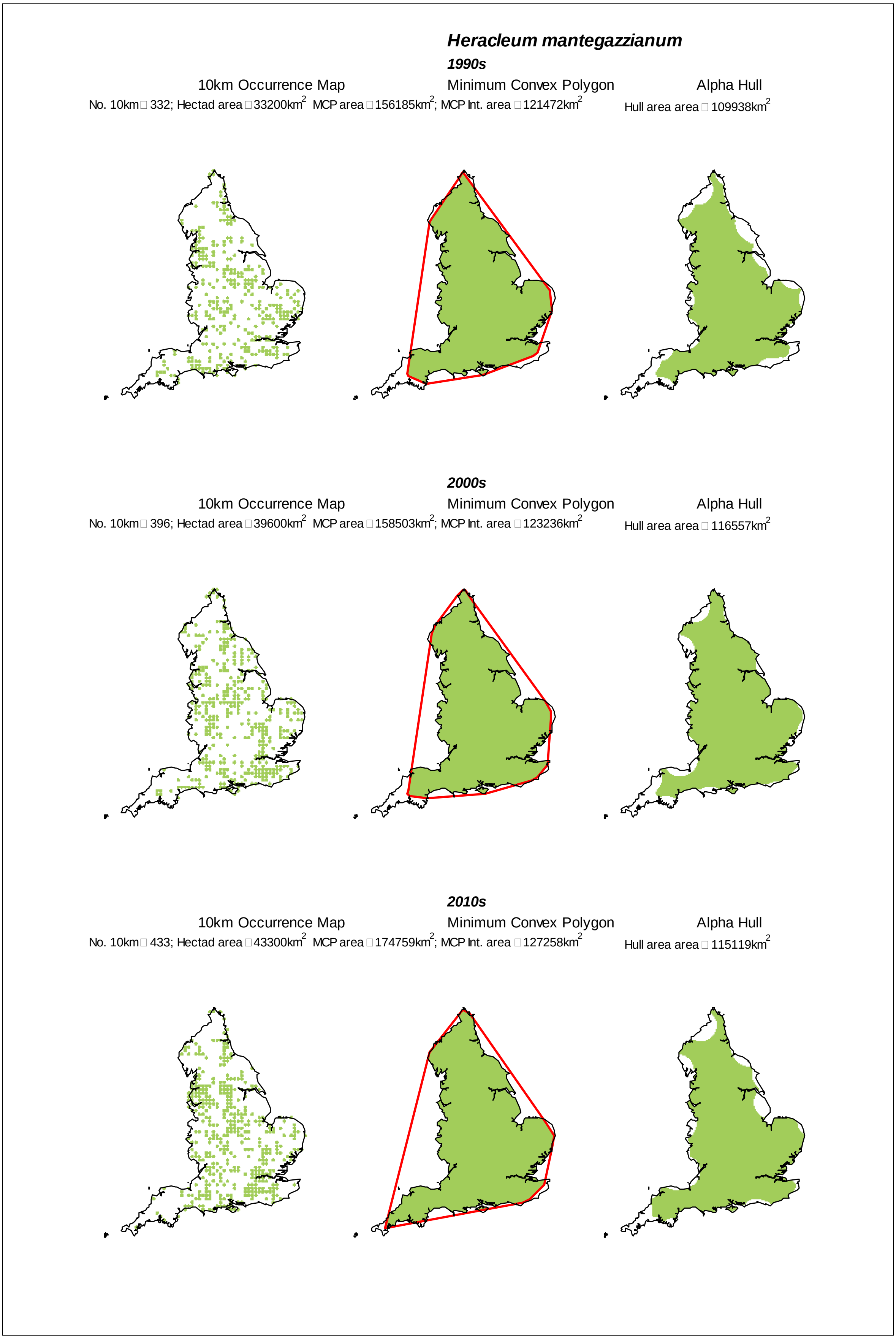


Figure C.35b Change in area of extent for *Heracleum mantegazzianum* by decade (1990s to 2010s)

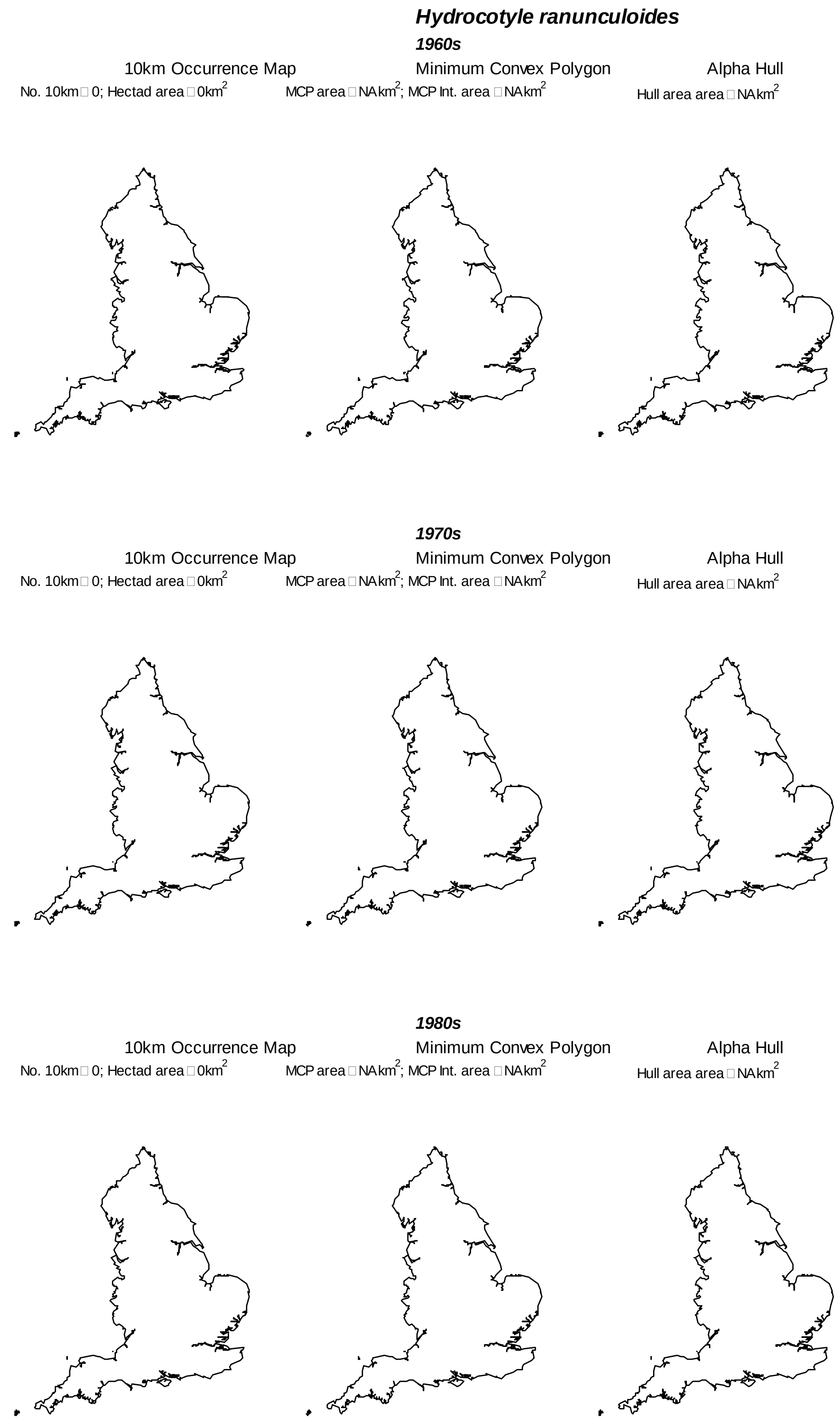


Figure C.36a Change in area of extent for *Hydrocotyle ranunculoides* by decade (1960s to 1980s)

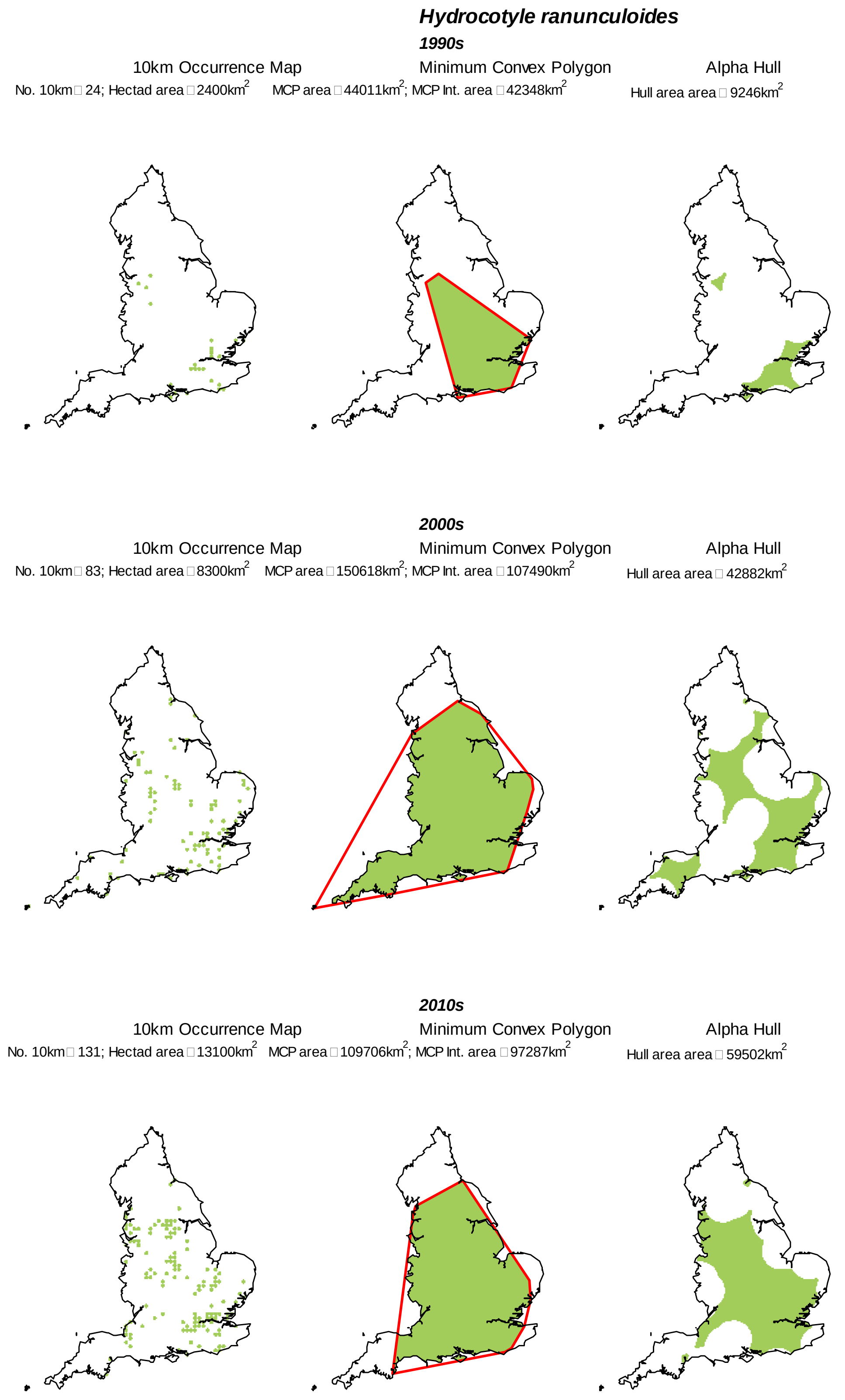


Figure C.36b Change in area of extent for *Hydrocotyle ranunculoides* by decade (1990s to 2010s)

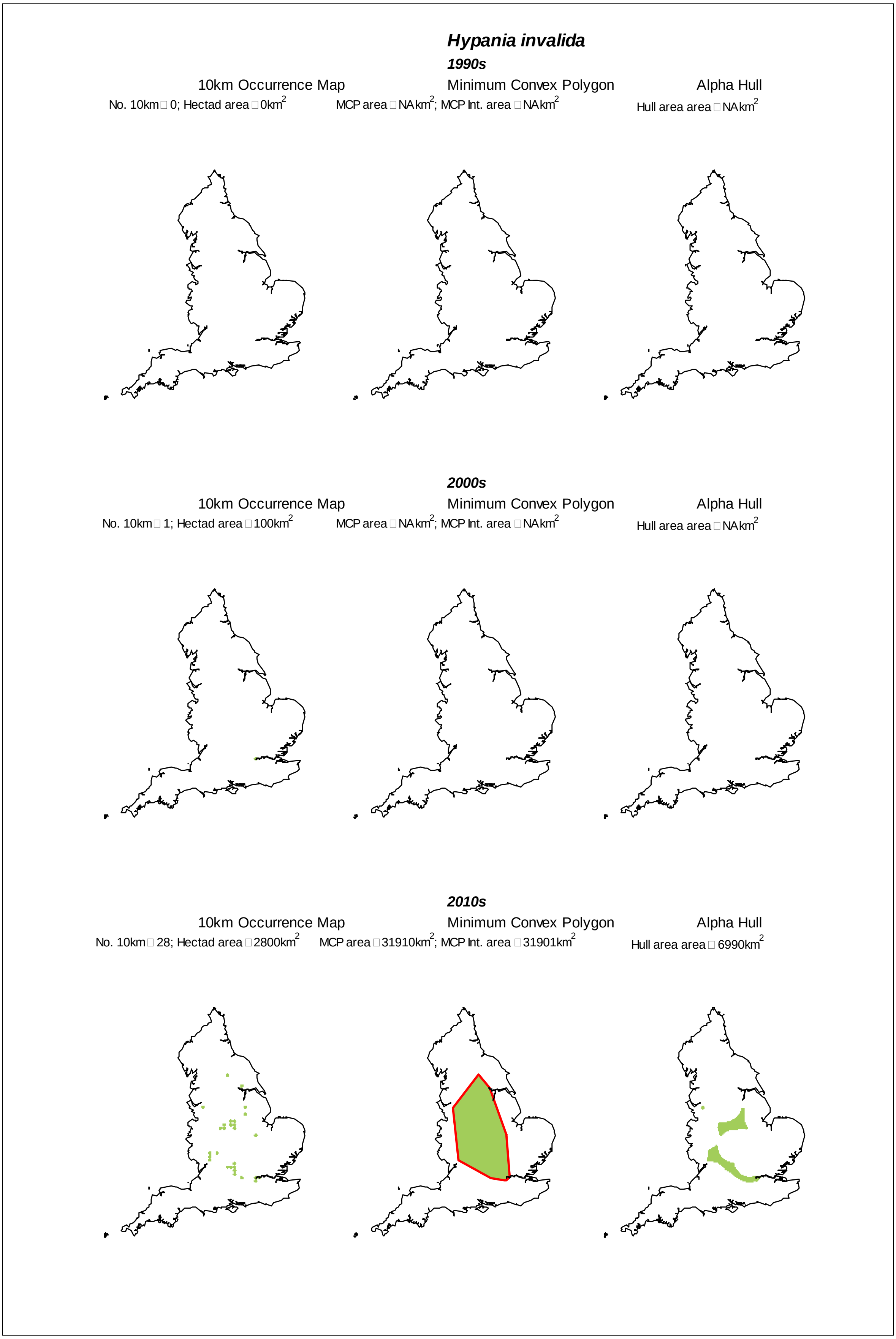


Figure C.37 Change in area of extent for *Hypania invalida* by decade (1990s to 2010s)

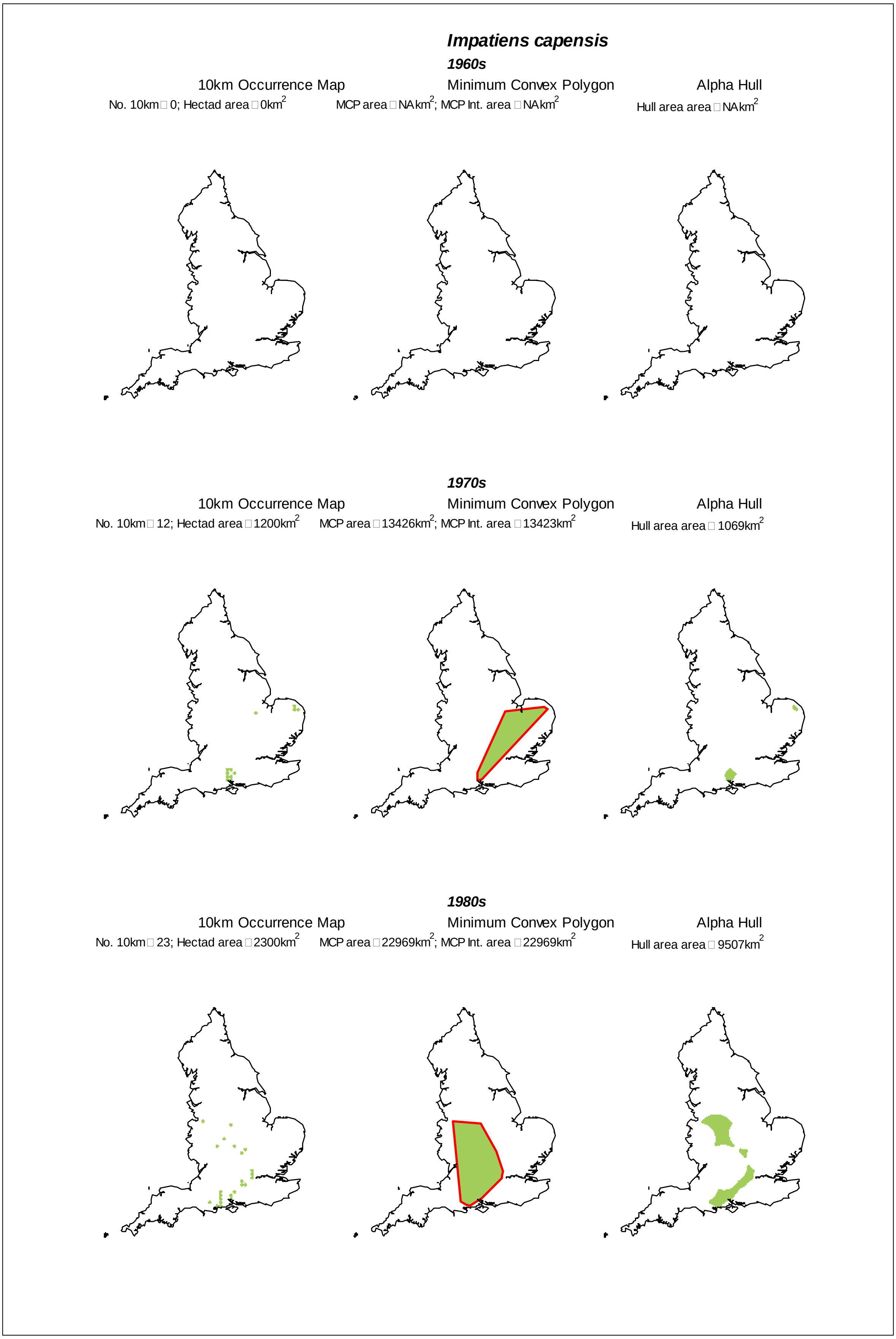


Figure C.38a Change in area of extent for *Impatiens capensis* by decade (1960s to 1980s)

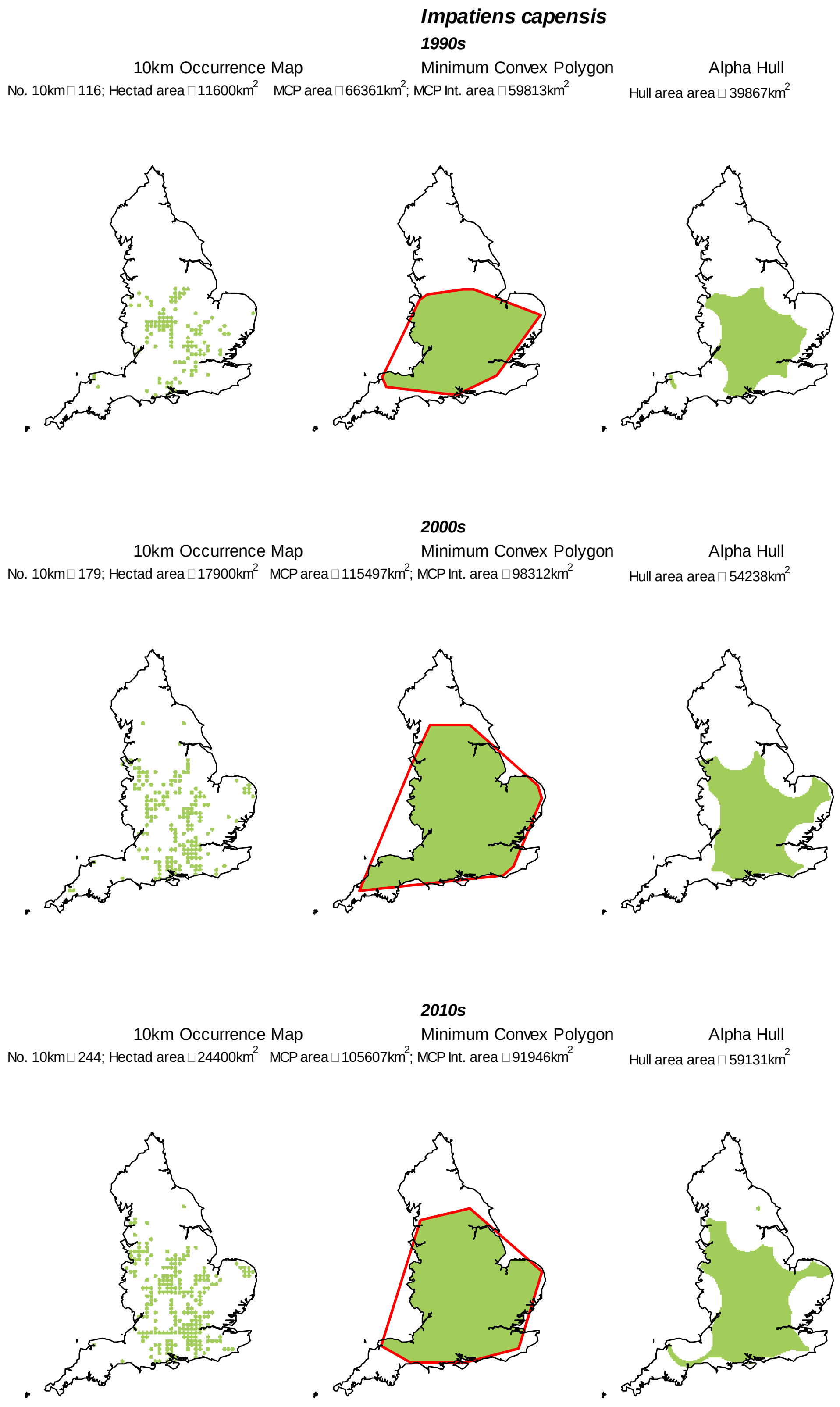


Figure C.38b Change in area of extent for *Impatiens capensis* by decade (1990s to 2010s)

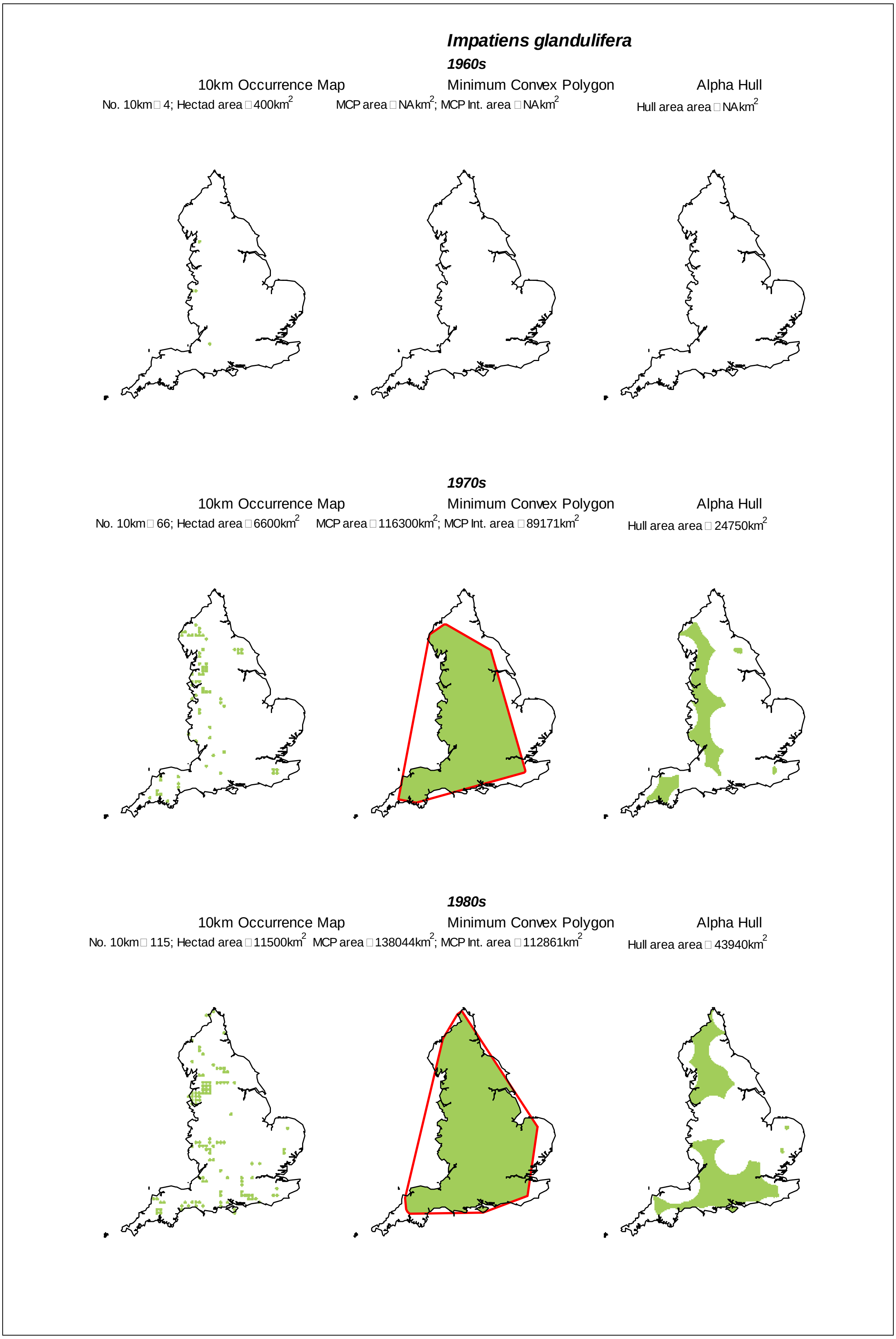


Figure C.39a Change in area of extent for *Impatiens glandulifera* by decade (1960s to 1980s)

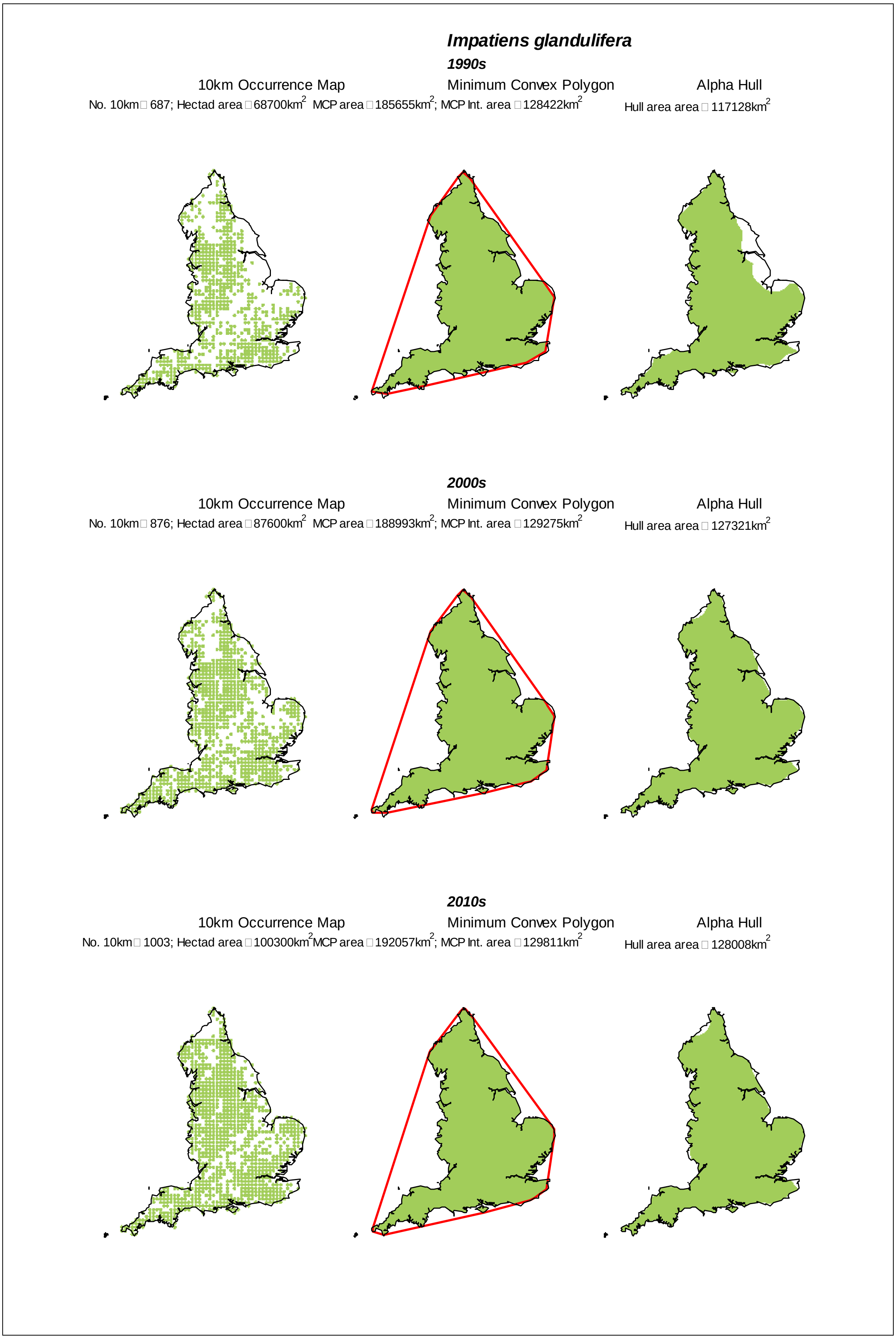


Figure C.39b Change in area of extent for *Impatiens glandulifera* by decade (1990s to 2010s)

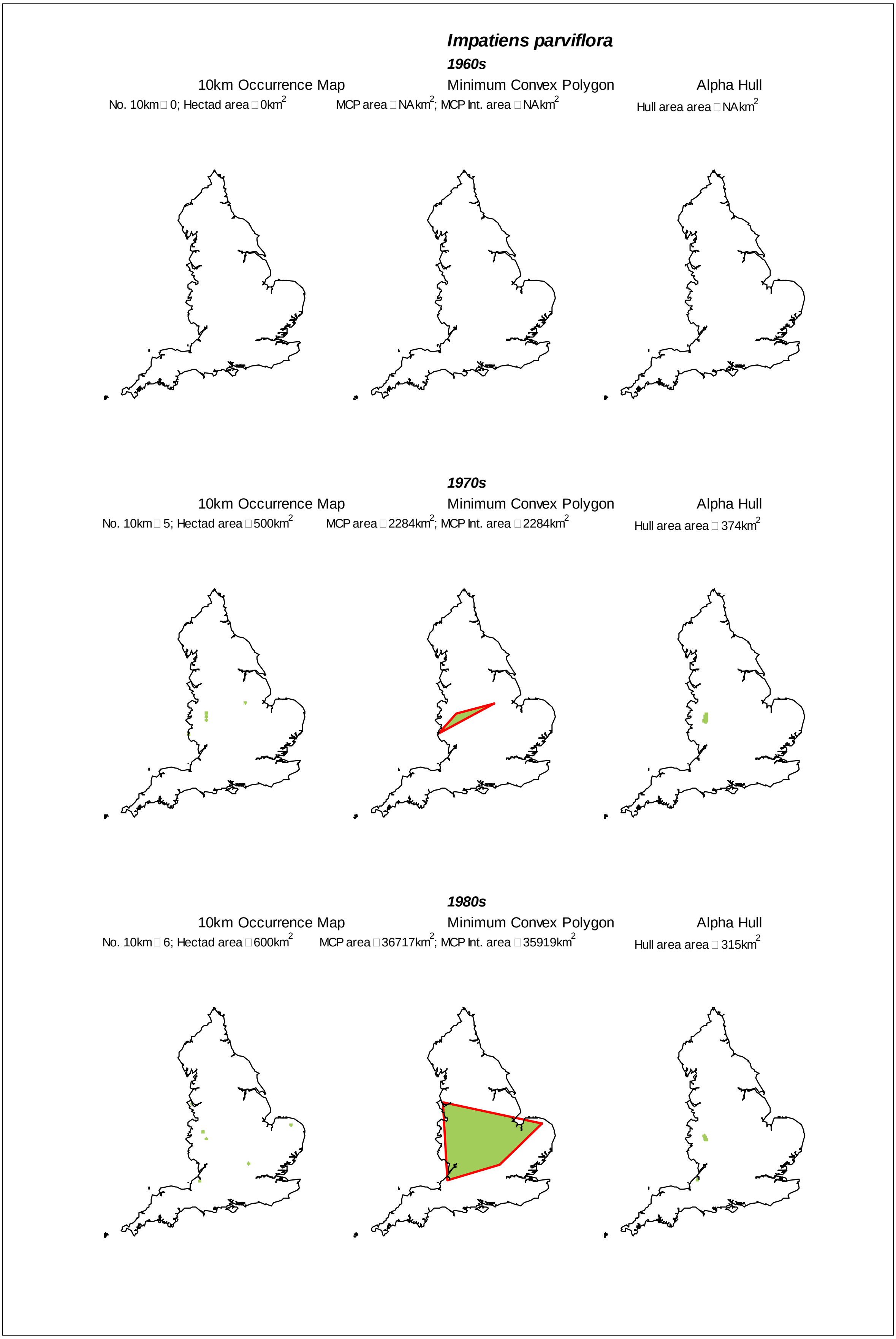


Figure C.40a Change in area of extent for *Impatiens parviflora* by decade (1960s to 1980s)

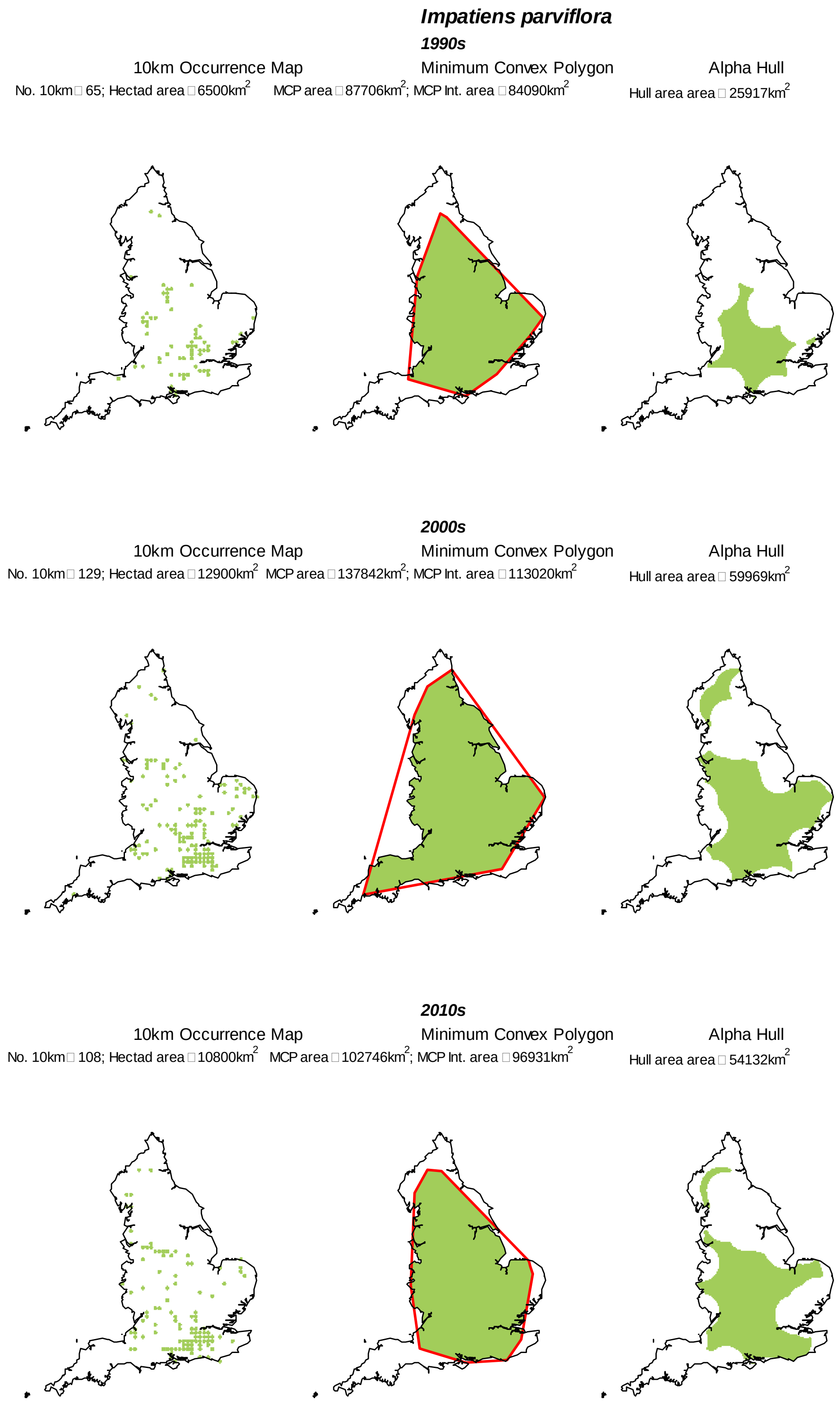


Figure C.40b Change in area of extent for *Impatiens parviflora* by decade (1990s to 2010s)

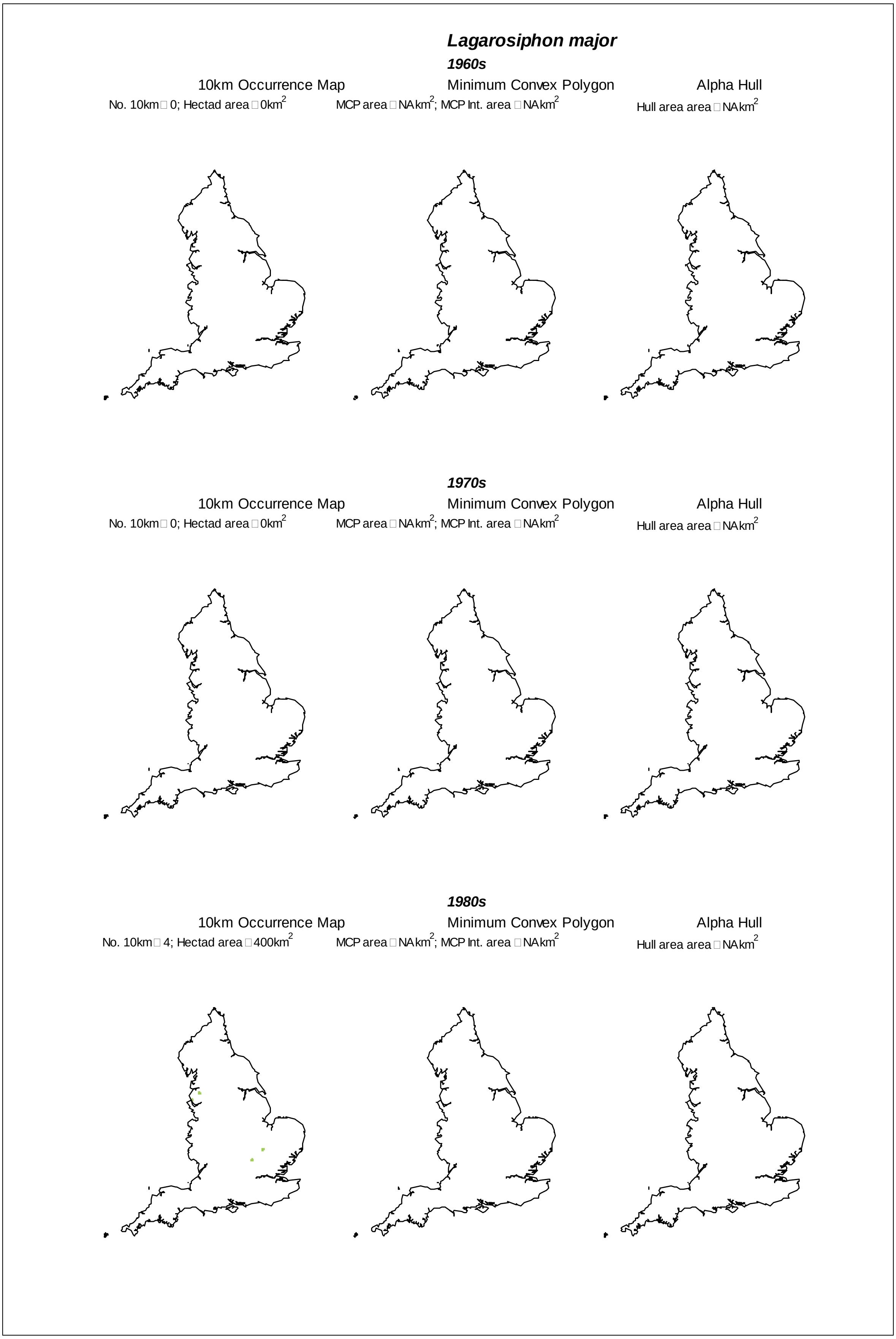


Figure C.41a Change in area of extent for *Lagarosiphon major* by decade (1960s to 1980s)

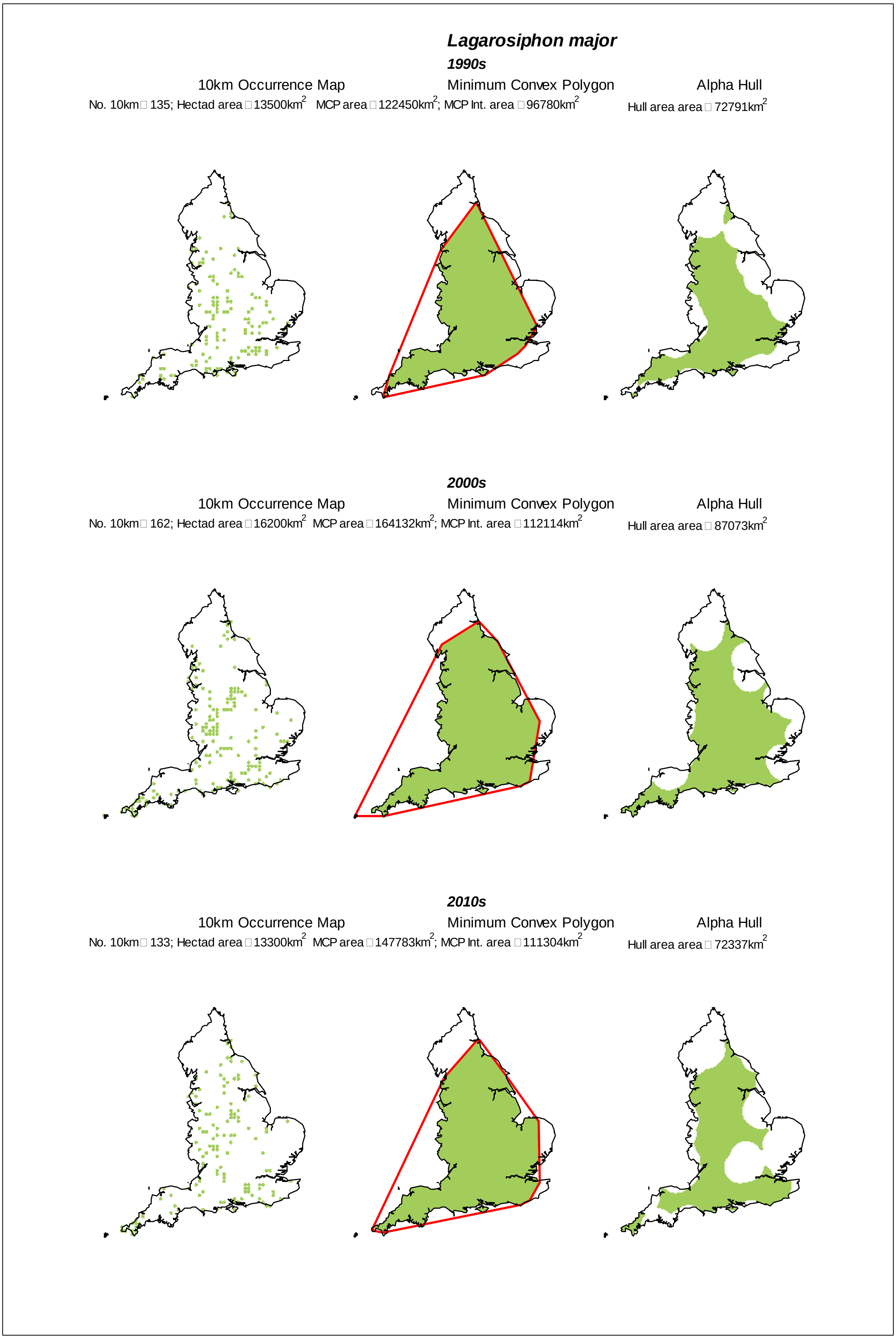


Figure C.41b Change in area of extent for *Lagarosiphon major* by decade (1990s to 2010s)

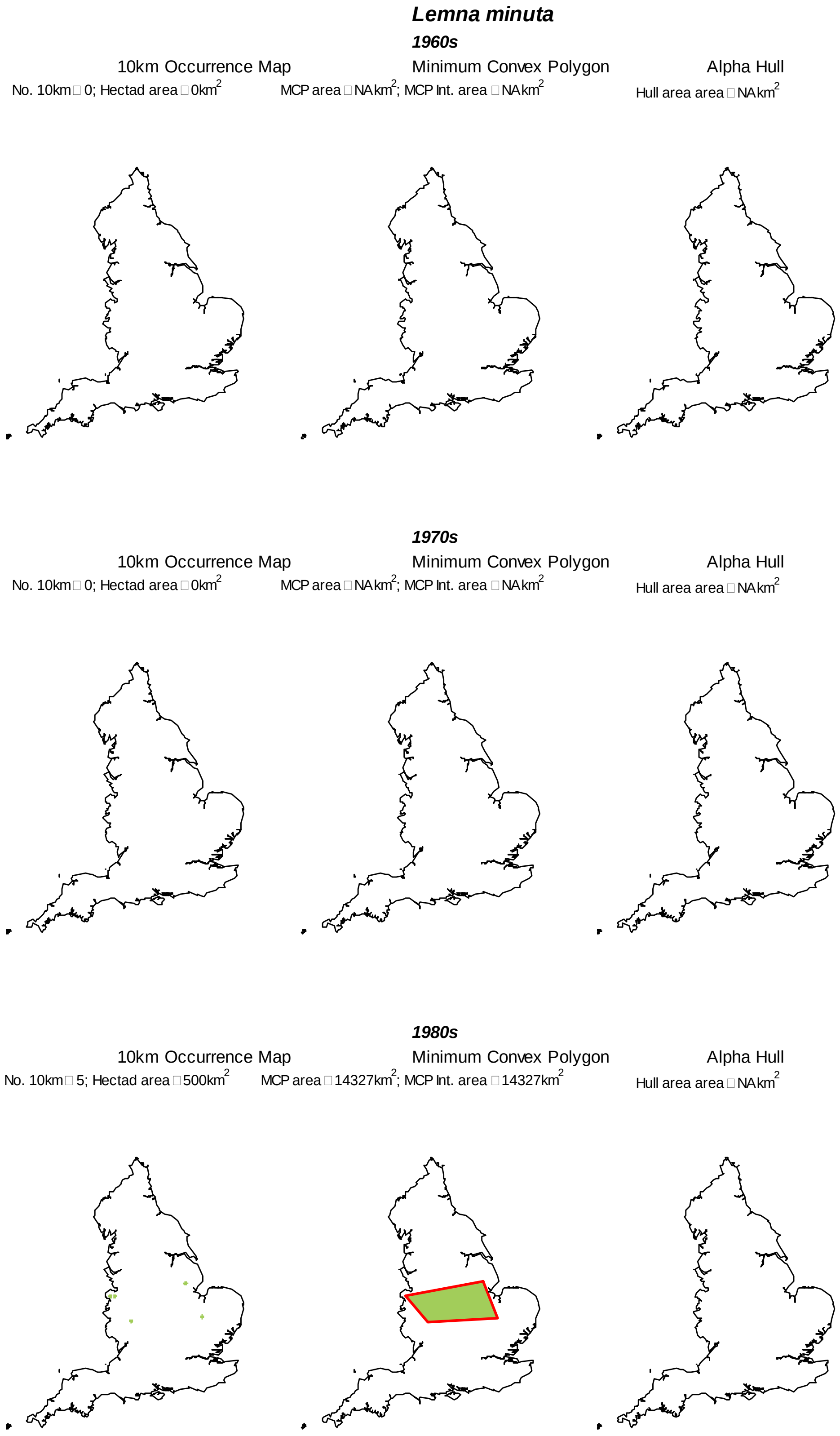
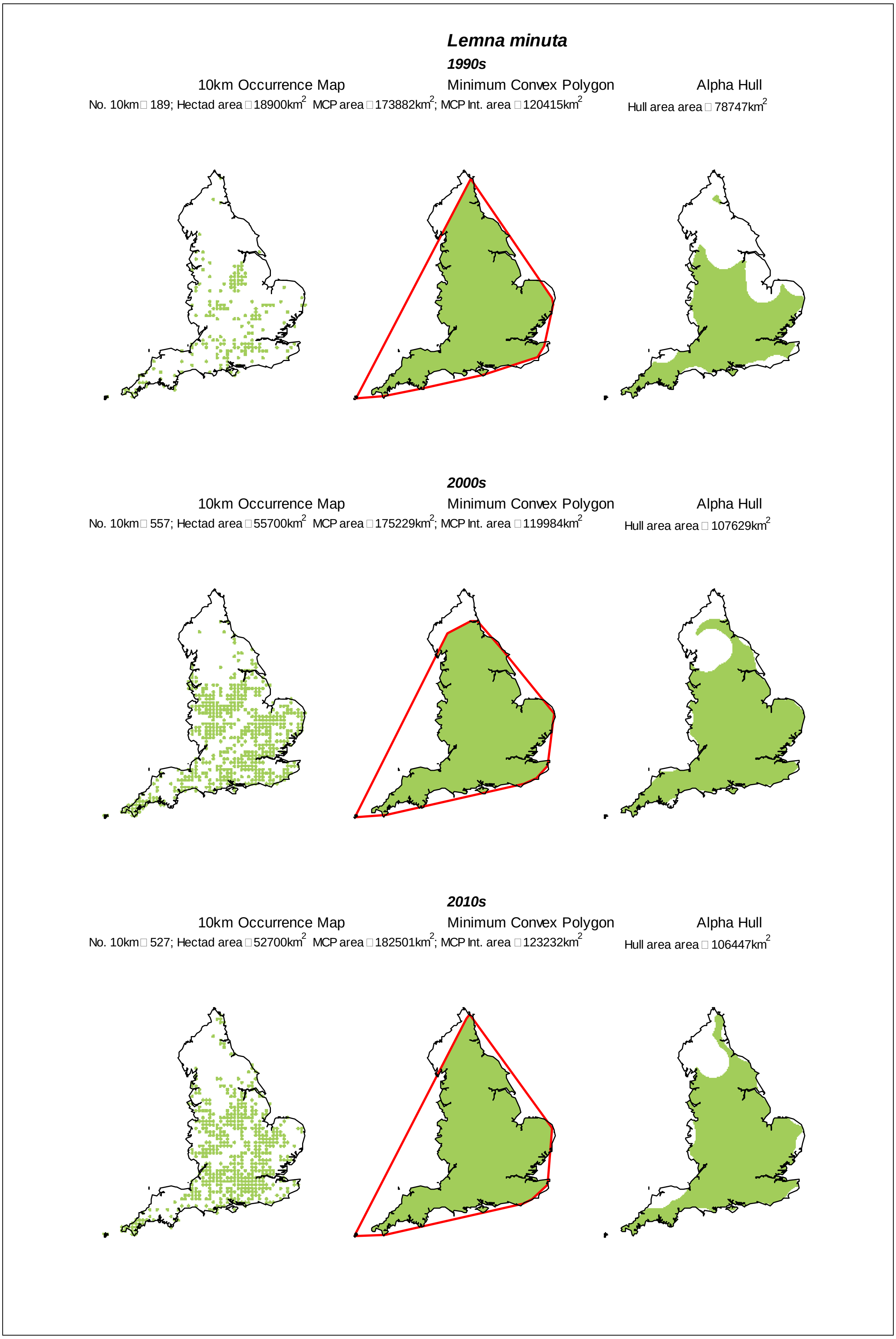


Figure C.42a Change in area of extent for *Lemna minuta* by decade (1960s to 1980s)



Lemna minuta

1990s

10km Occurrence Map — Minimum Convex Polygon — Alpha Hull

No. 10km□ 189; Hectad area□18900km^2 MCP area□173882km^2; MCP Int. area □120415km^2

Hull area area□ 78747km^2

2000s

10km Occurrence Map — Minimum Convex Polygon — Alpha Hull

No. 10km□ 557; Hectad area□55700km^2 MCP area□175229km^2; MCP Int. area □119984km^2

Hull area area□ 107629km^2

2010s

10km Occurrence Map — Minimum Convex Polygon — Alpha Hull

No. 10km□ 527; Hectad area□52700km^2 MCP area□182501km^2; MCP Int. area □123232km^2

Hull area area□ 106447km^2

Figure C.42b Change in area of extent for *Lemna minuta* by decade (1990s to 2010s)

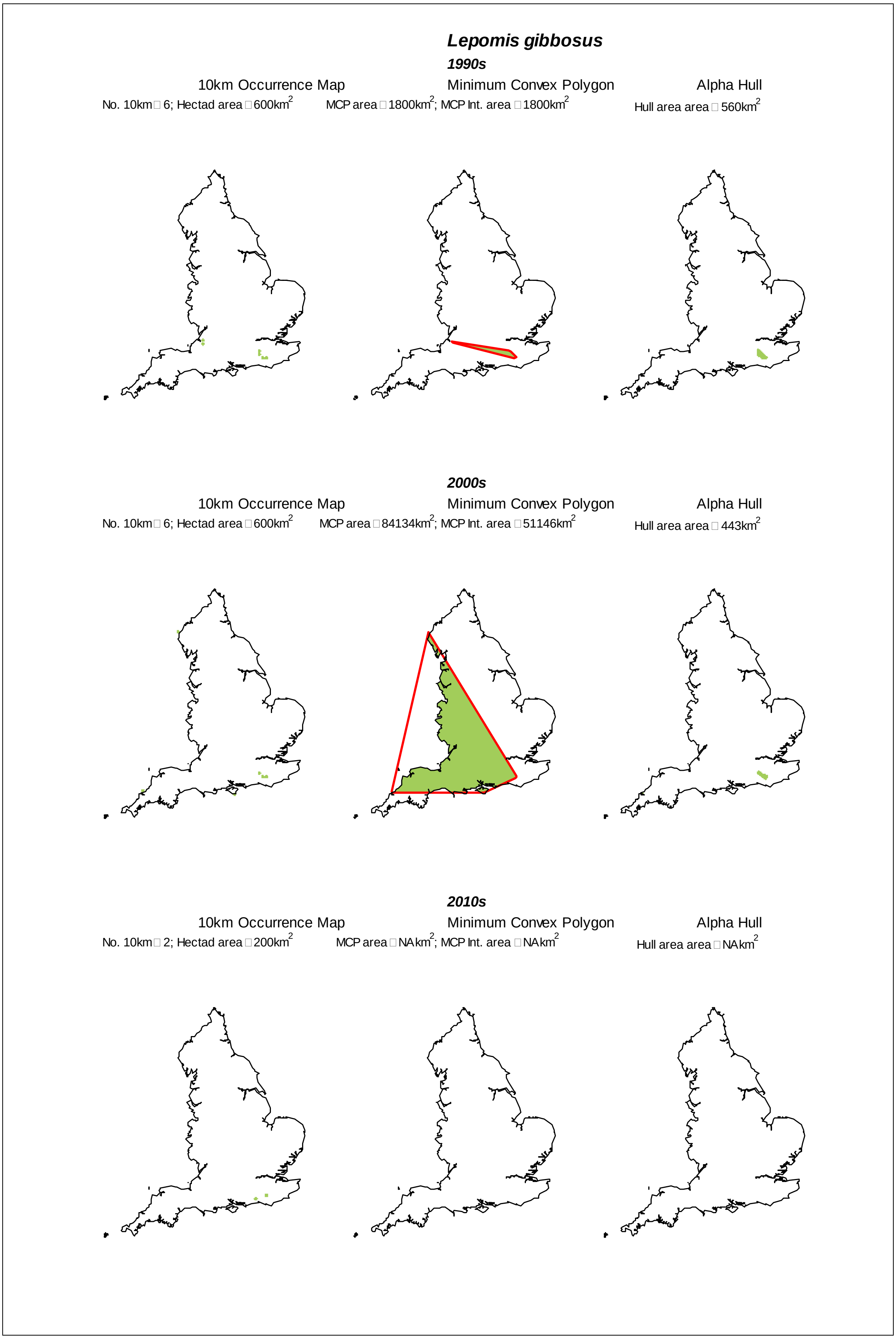


Figure C.43 Change in area of extent for *Lepomis gibbosus* by decade (1990s to 2010s)

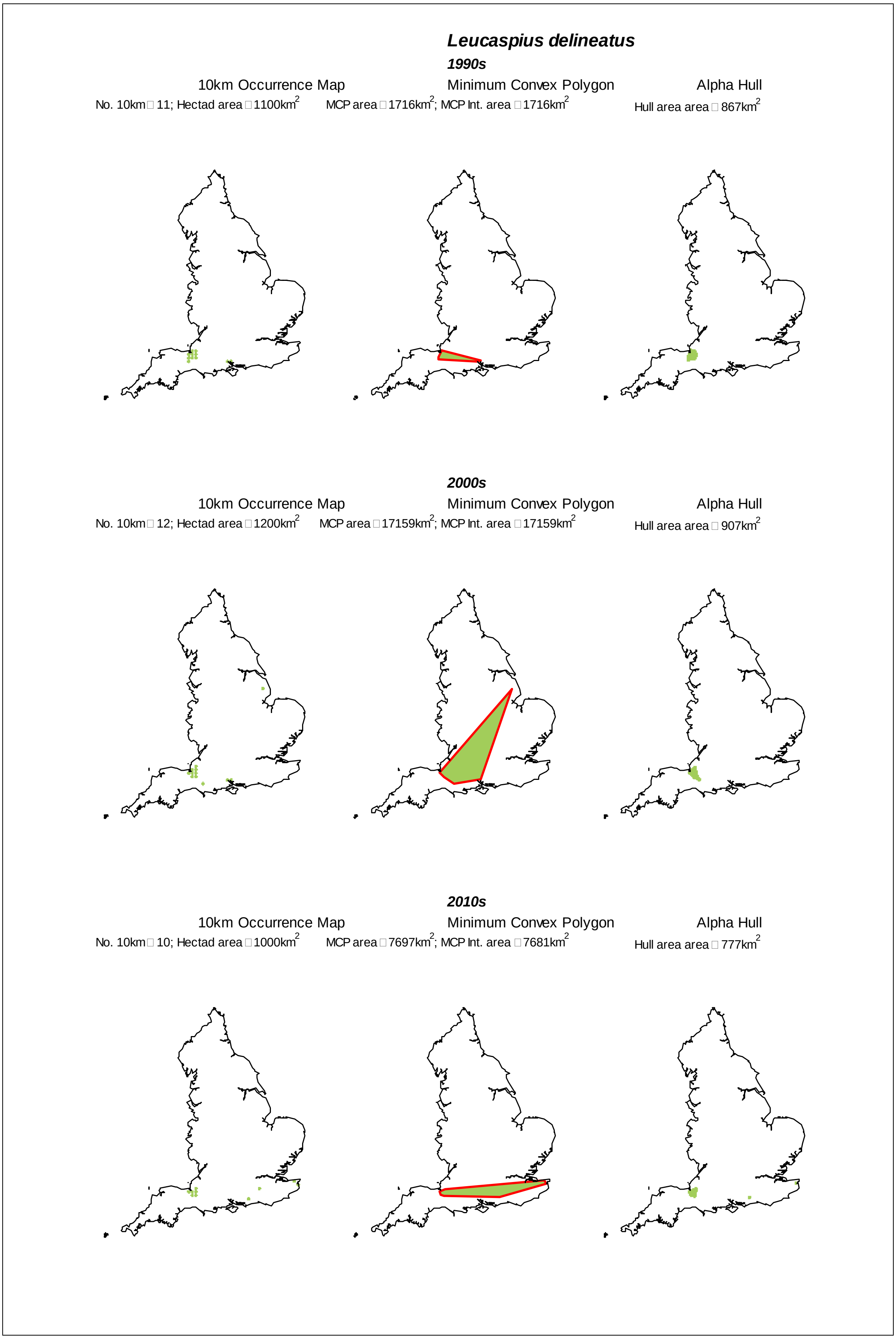


Figure C.44 Change in area of extent for *Leucaspius delineatus* by decade (1990s to 2010s)

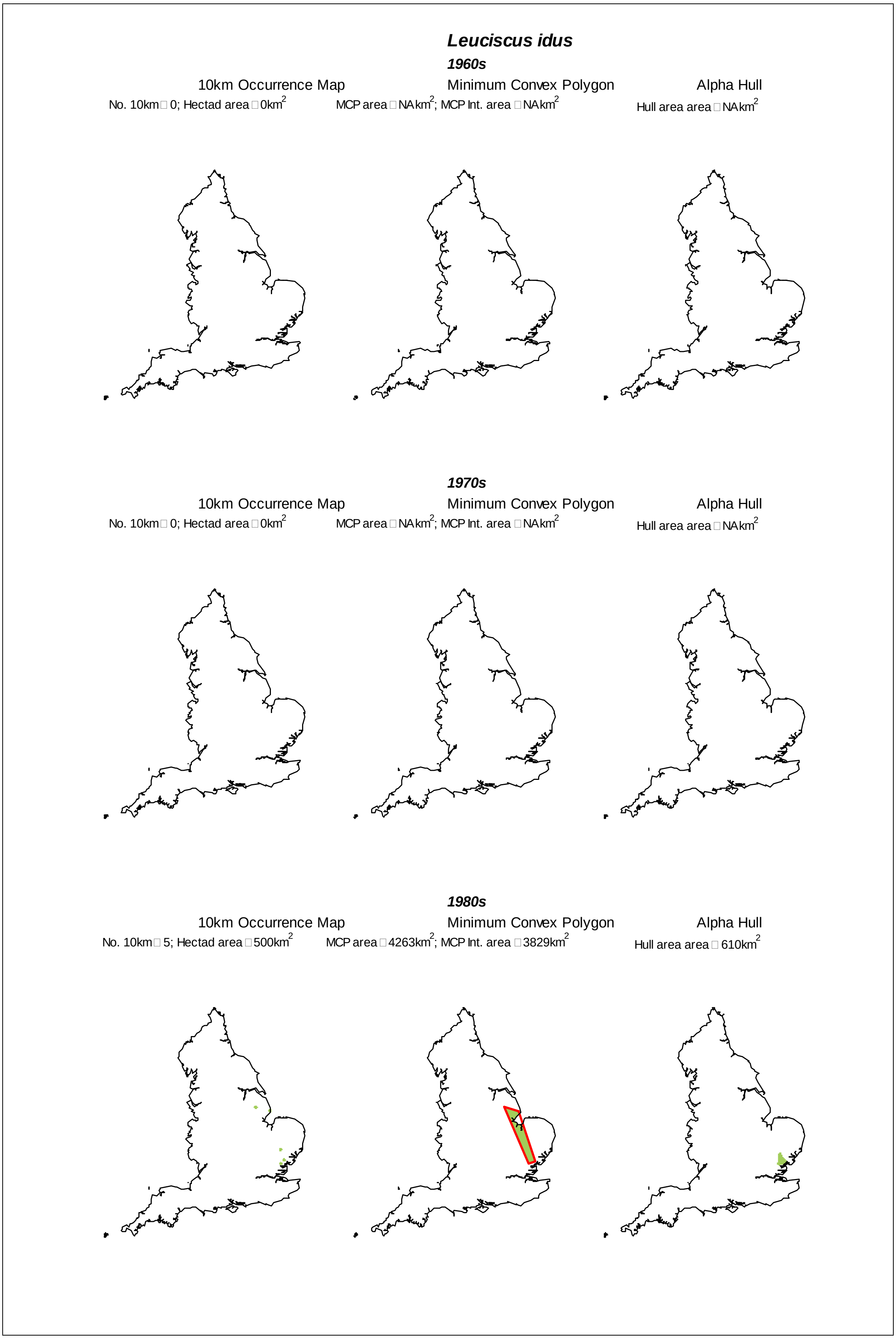


Figure C.45a Change in area of extent for *Leuciscus idus* by decade (1960s to 1980s)

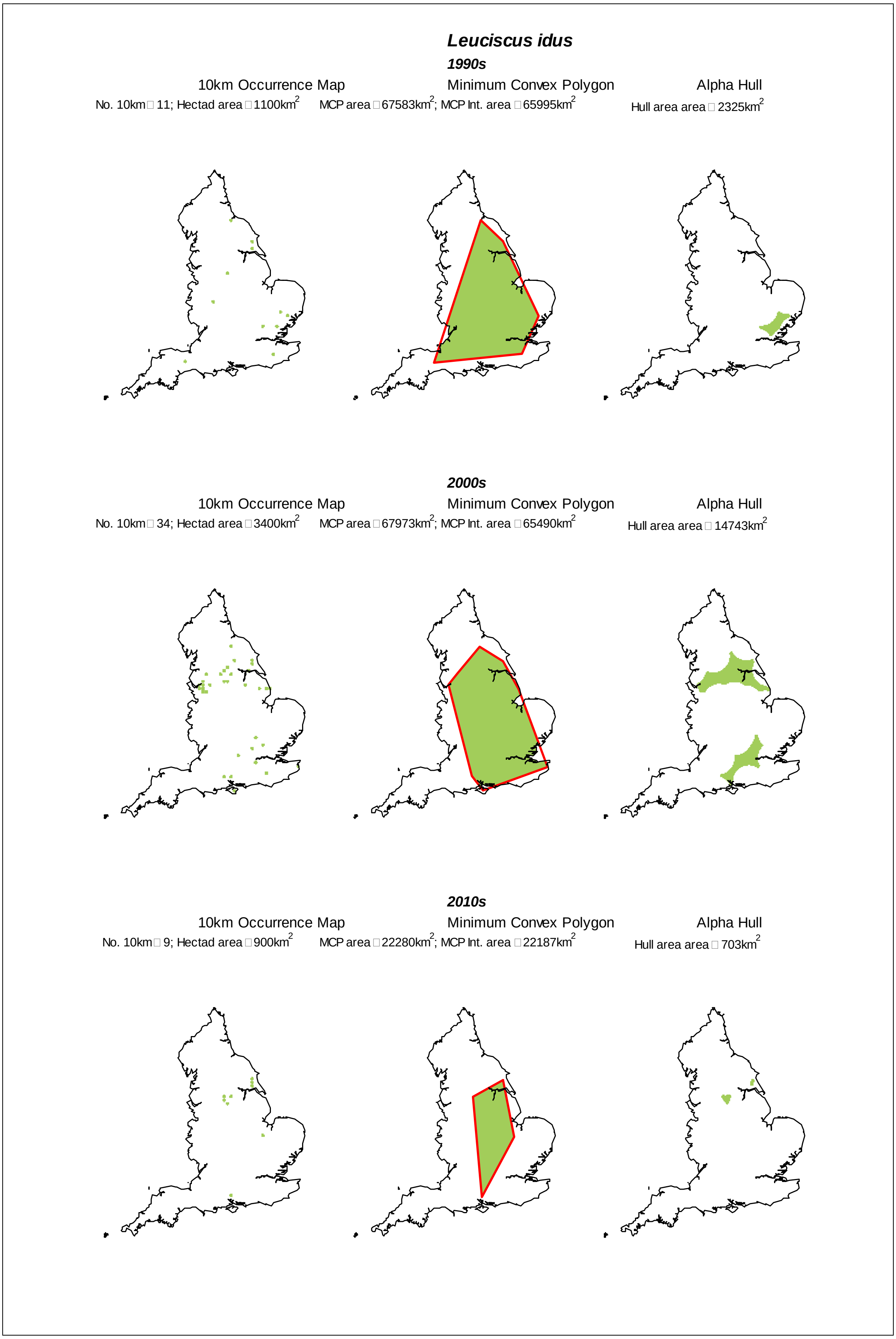


Figure C.45b Change in area of extent for *Leuciscus idus* by decade (1990s to 2010s)

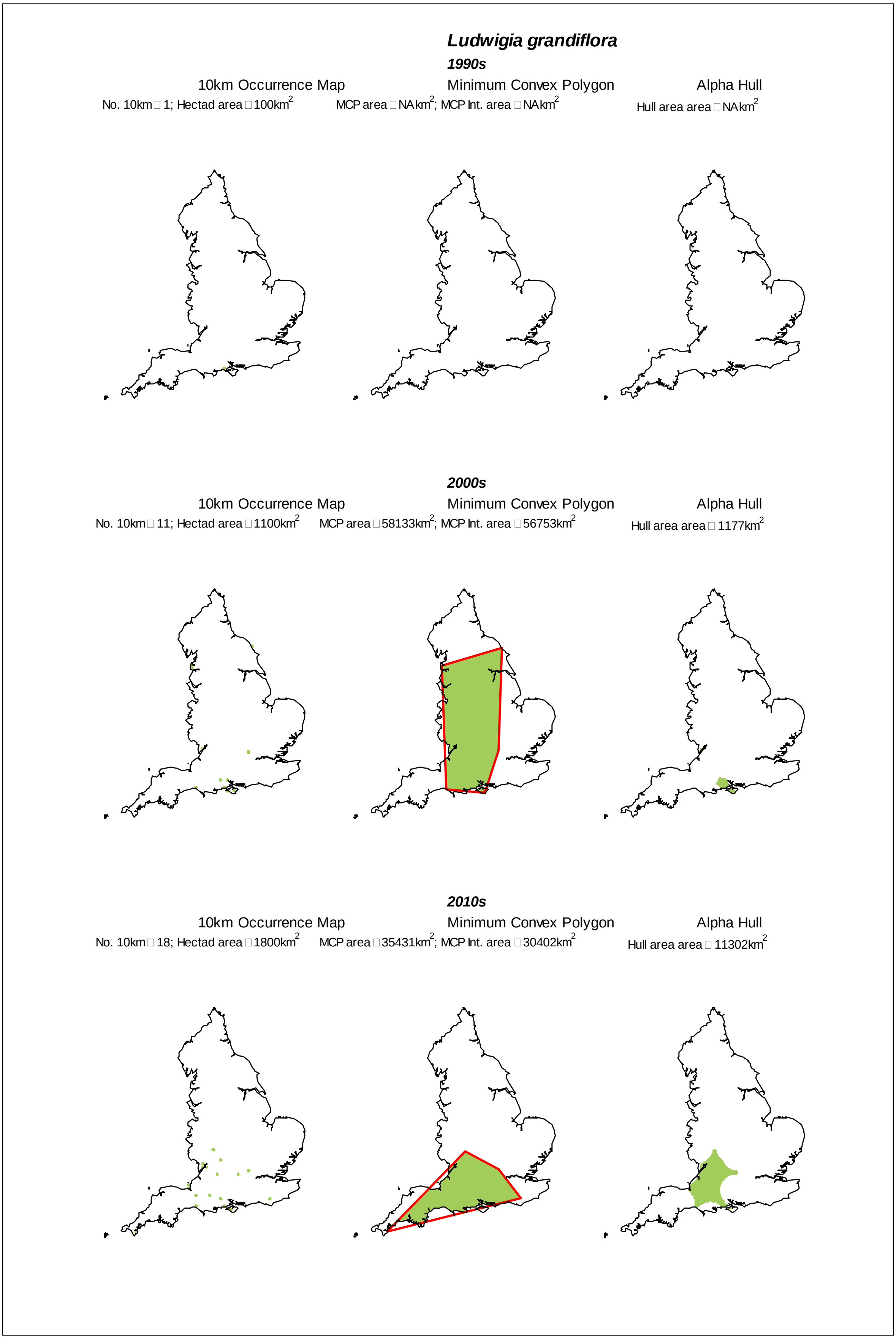


Figure C.46 Change in area of extent for *Ludwigia grandiflora* by decade (1990s to 2010s)

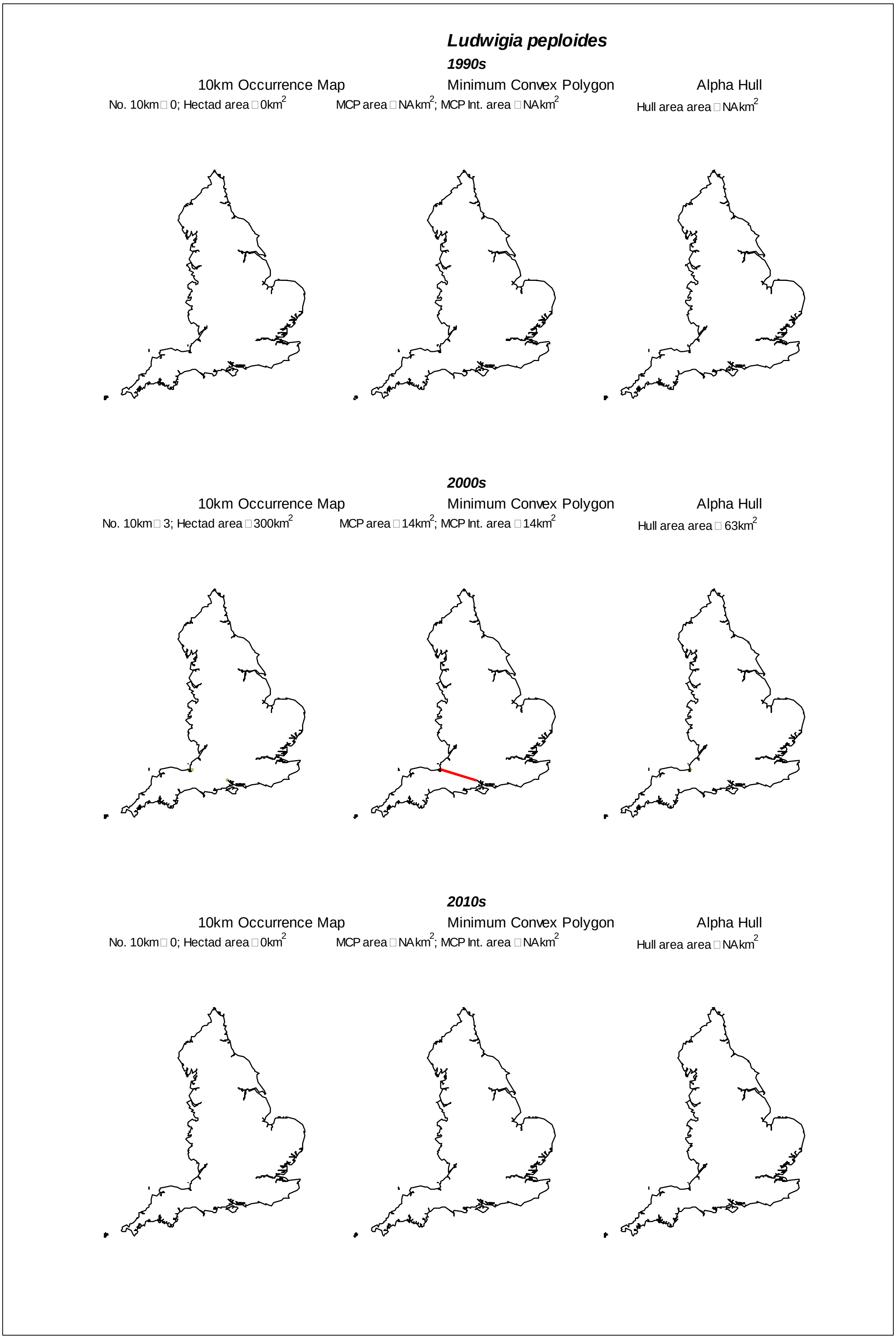


Figure C.47 Change in area of extent for *Ludwigia peploides* by decade (1990s to 2010s)

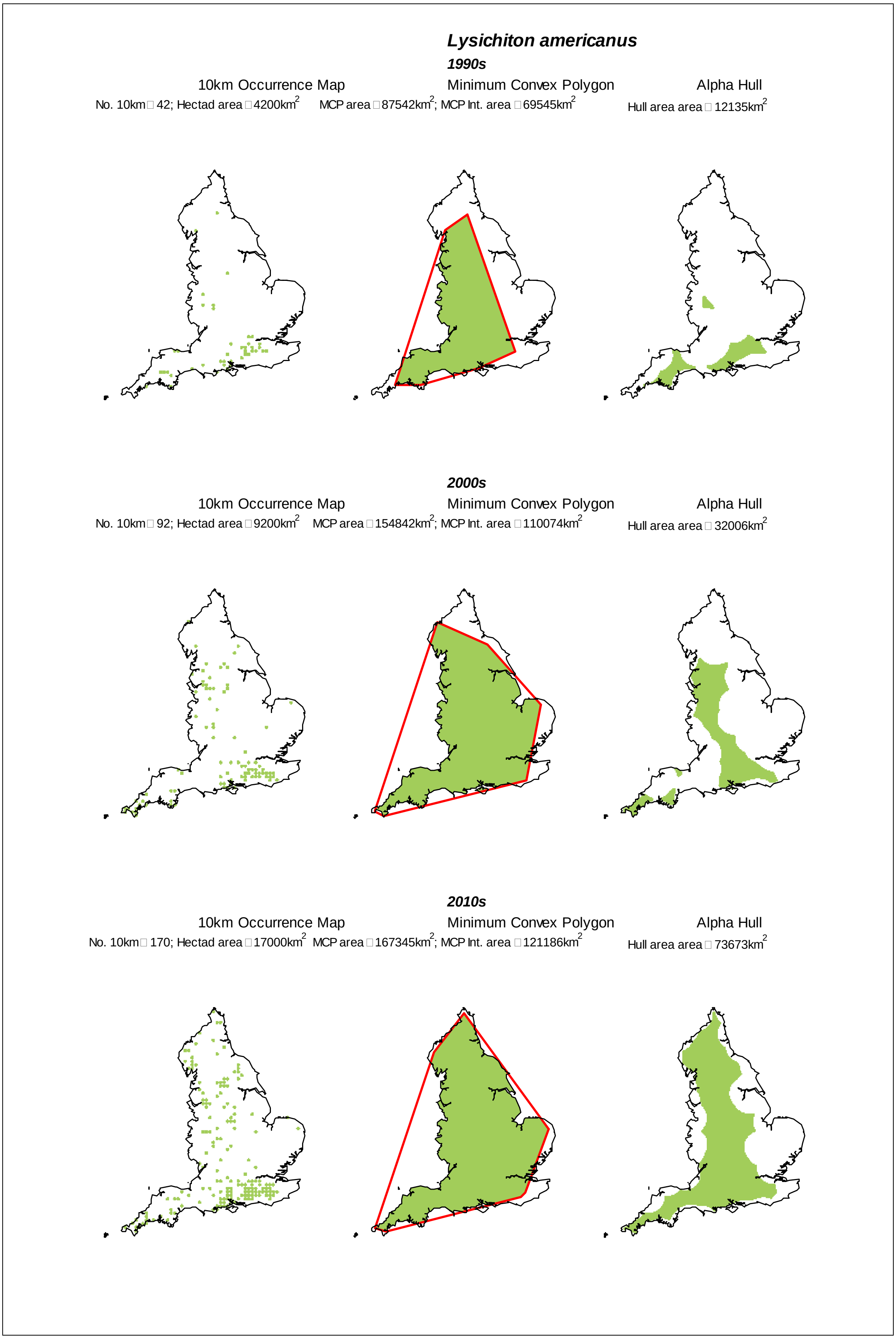


Figure C.48 Change in area of extent for *Lysichiton americanus* by decade (1990s to 2010s)

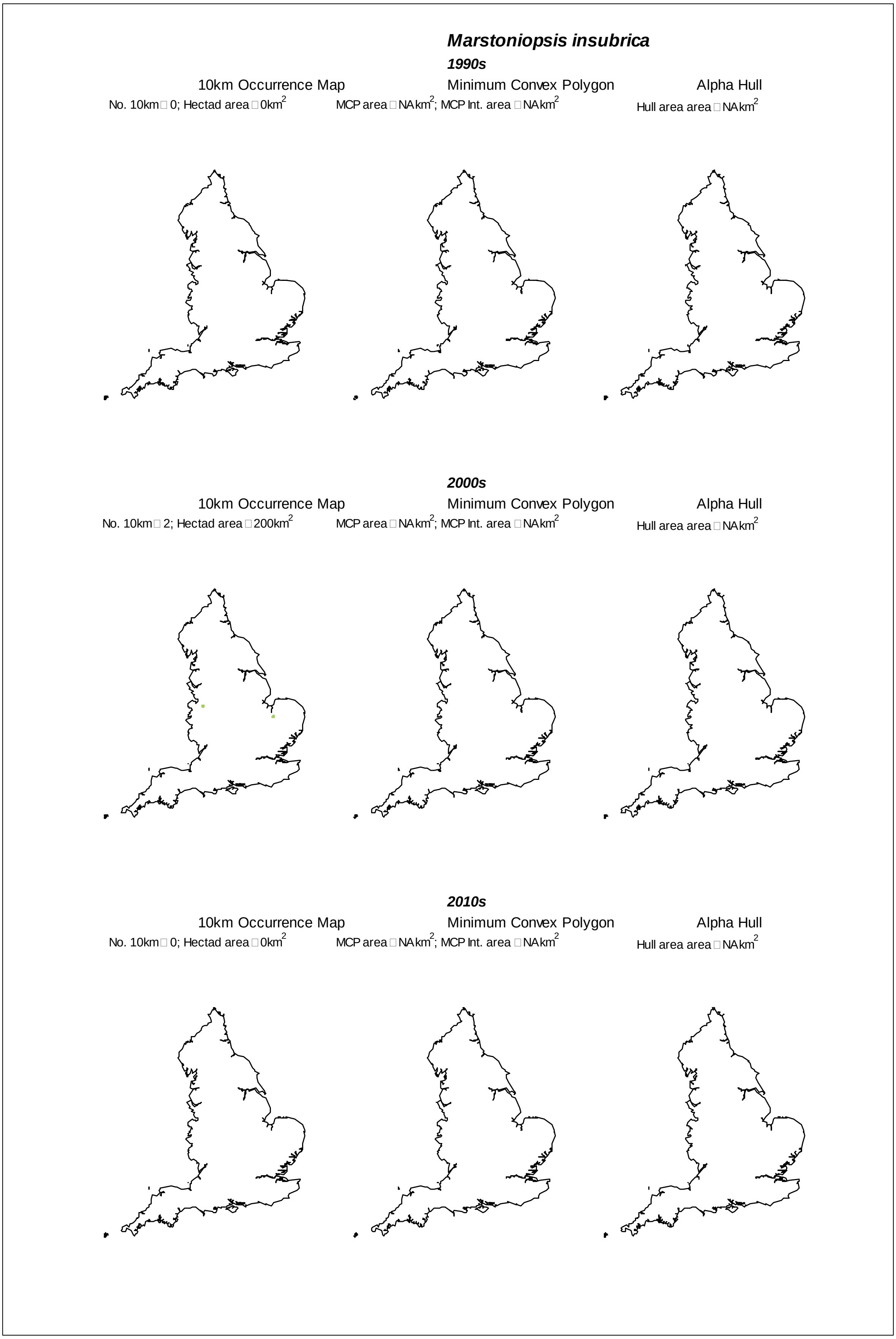


Figure C.49 Change in area of extent for *Marstoniopsis insubrica* by decade (1990s to 2010s)

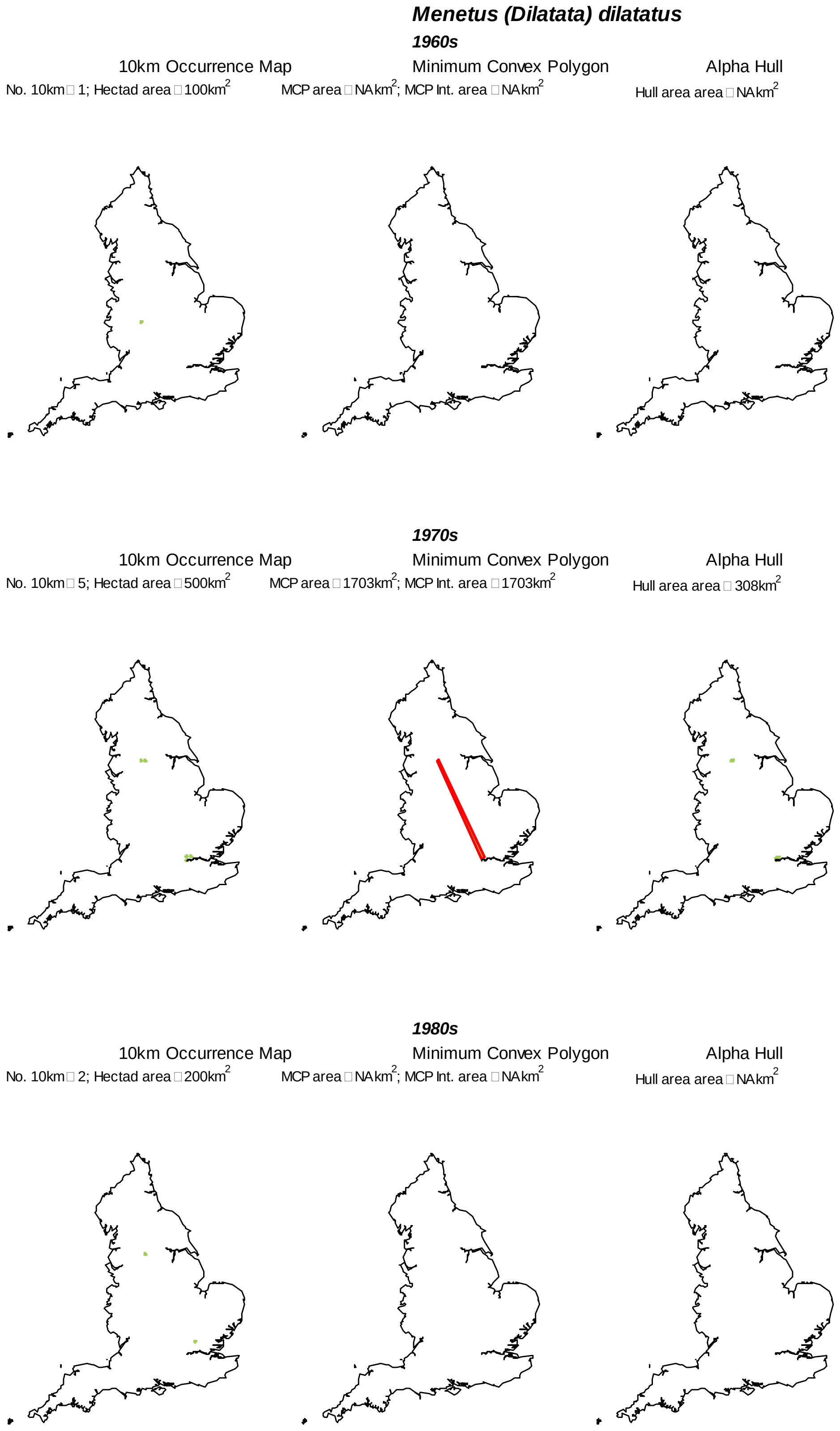


Figure C.50 Change in area of extent for *Menetus (Dilatata) dilatatus* by decade (1960s to 1980s)

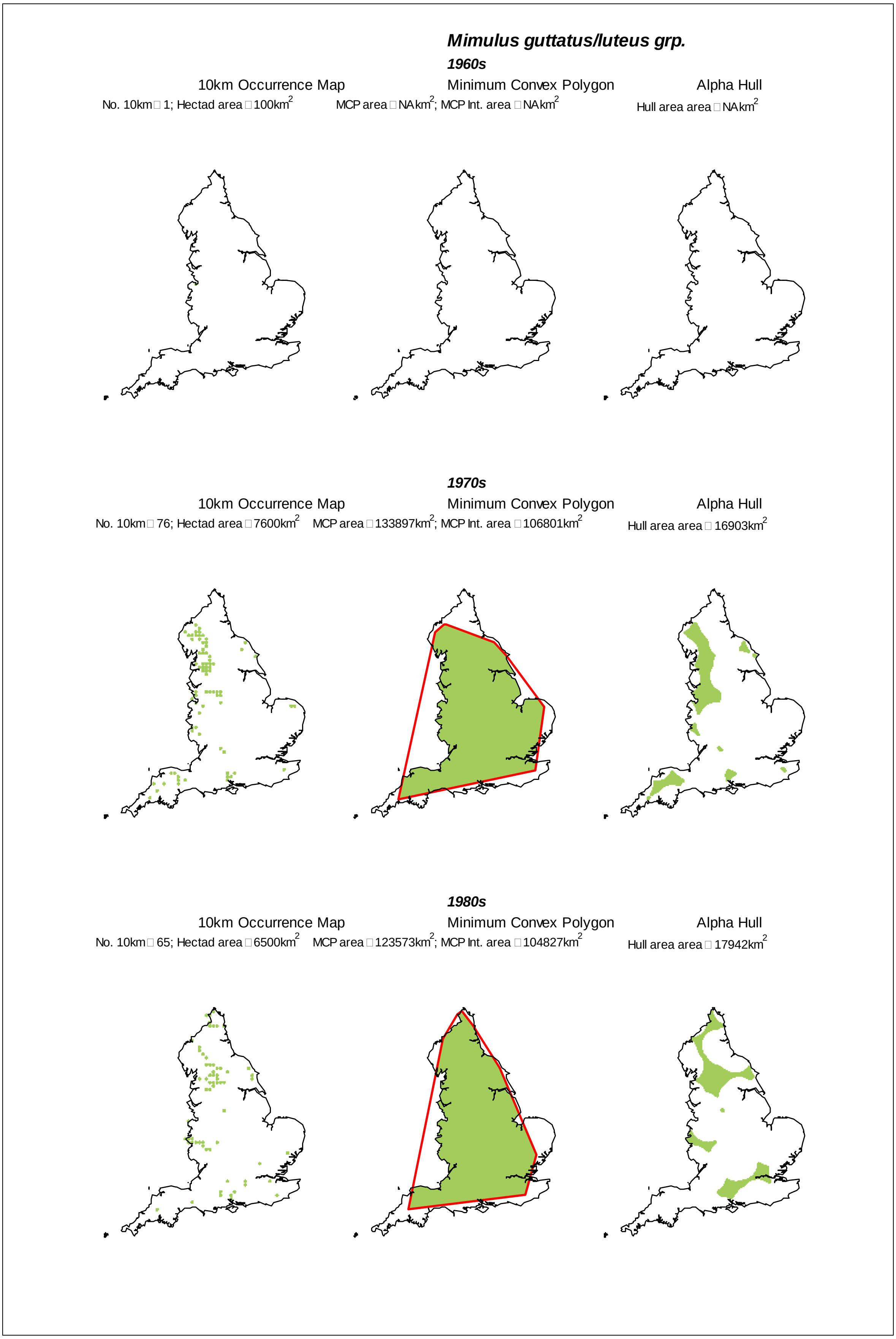


Figure C.51a Change in area of extent for *Mimulus guttatus/luteus* grp by decade (1960s to 1980s)

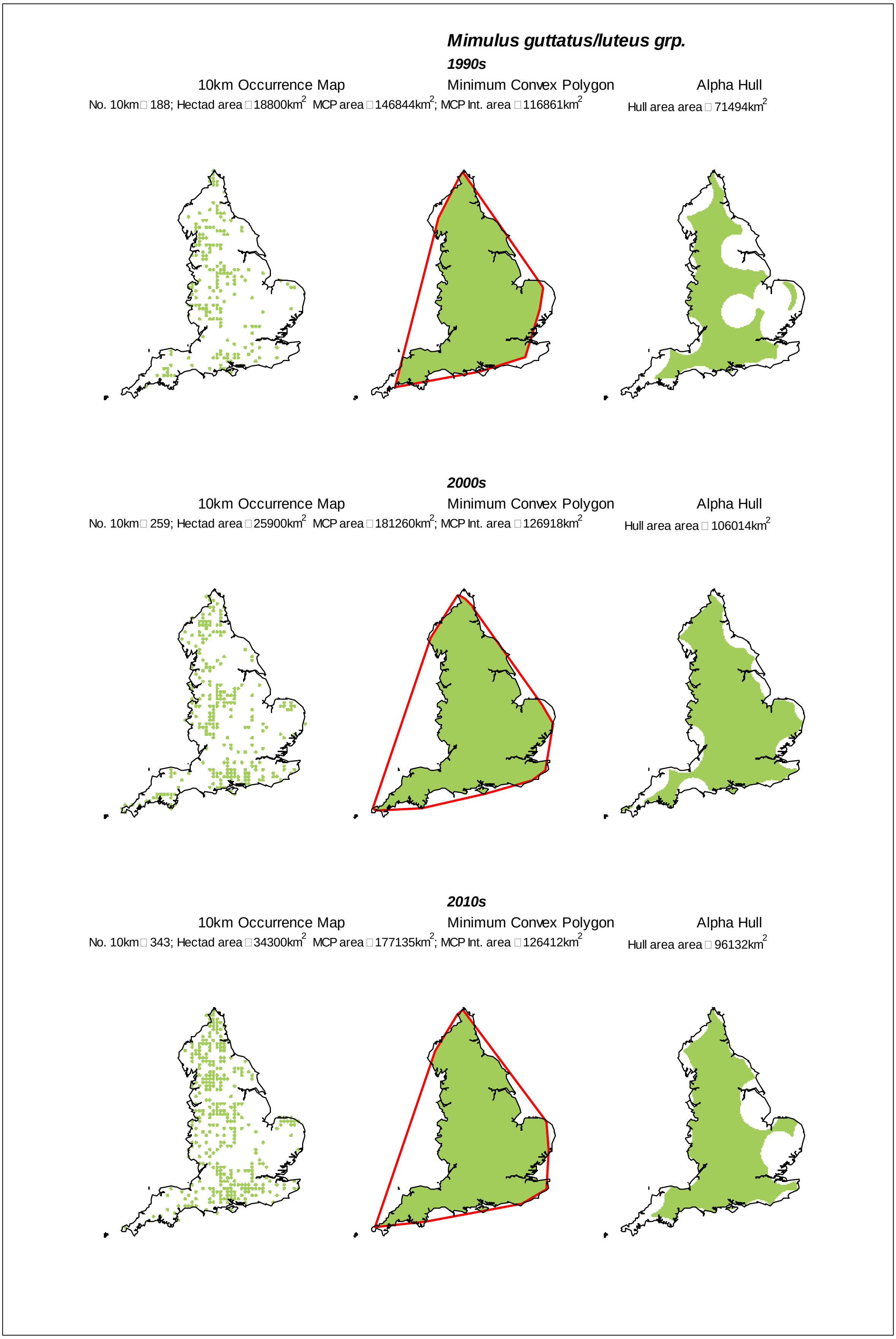


Figure C.51b Change in area of extent for *Mimulus guttatus/luteus* grp by decade (1990s to 2010s)

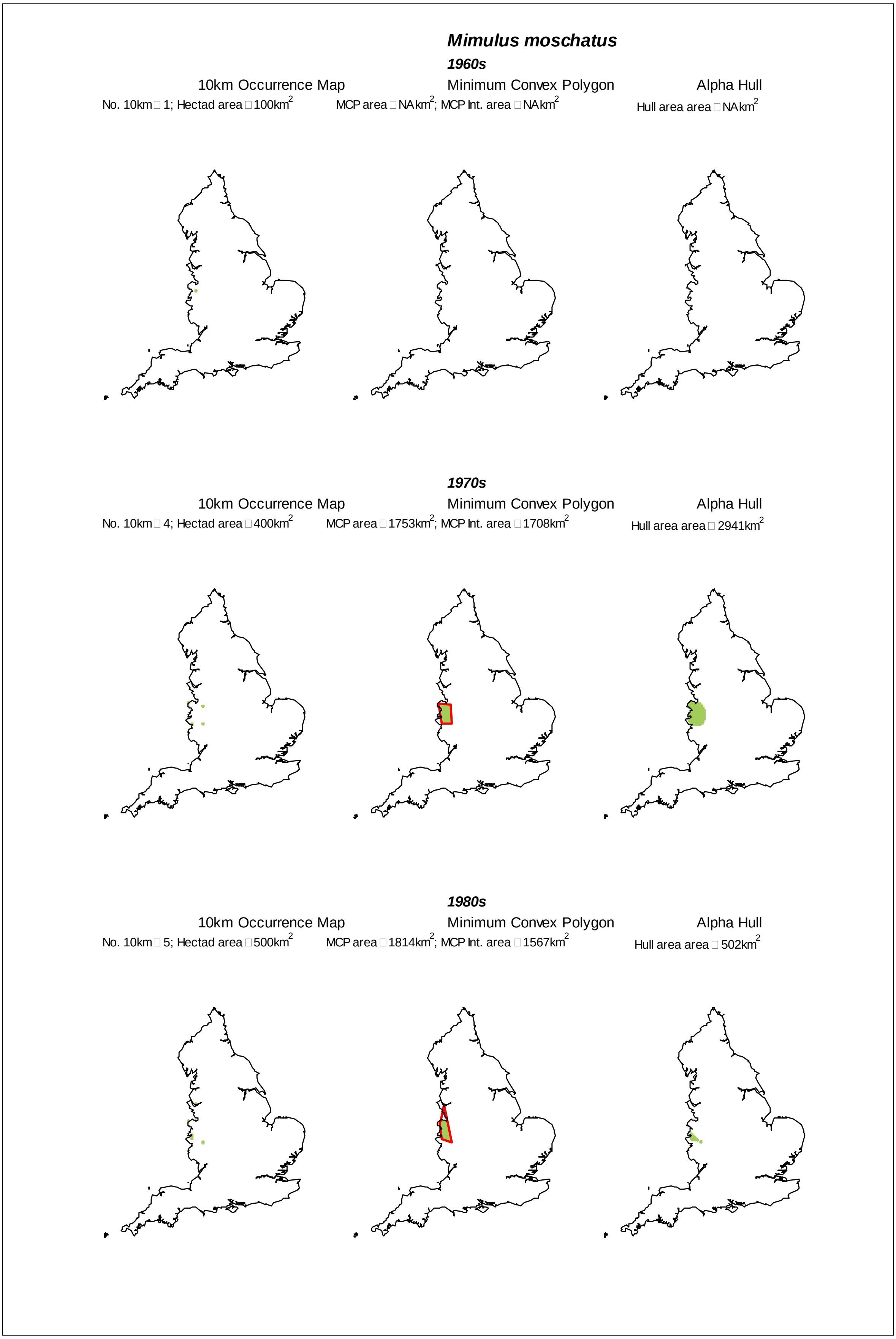


Figure C.52a Change in area of extent for *Mimulus moschatus* by decade (1960s to 1980s)

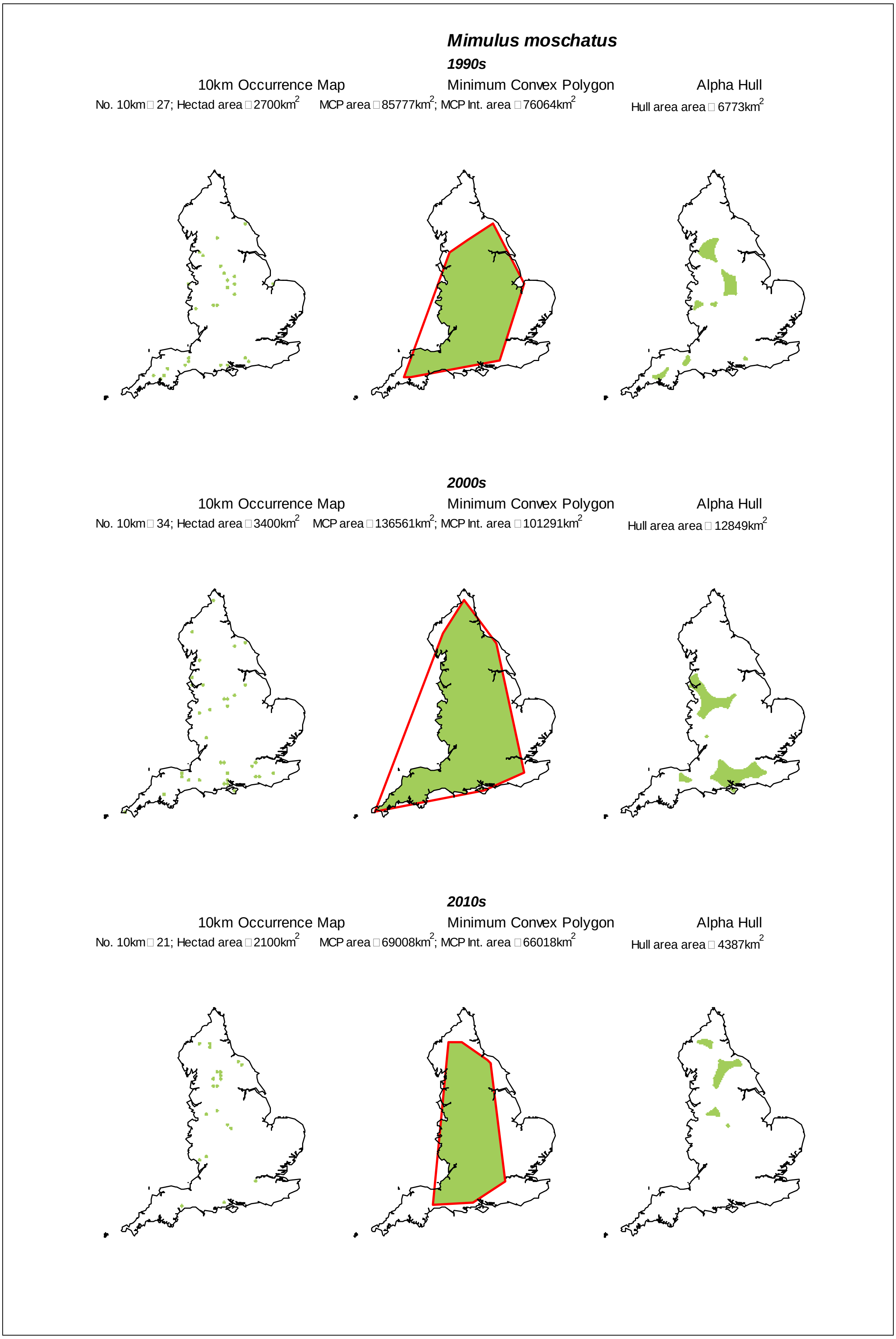


Figure C.52b Change in area of extent for *Mimulus moschatus* by decade (1990s to 2010s)

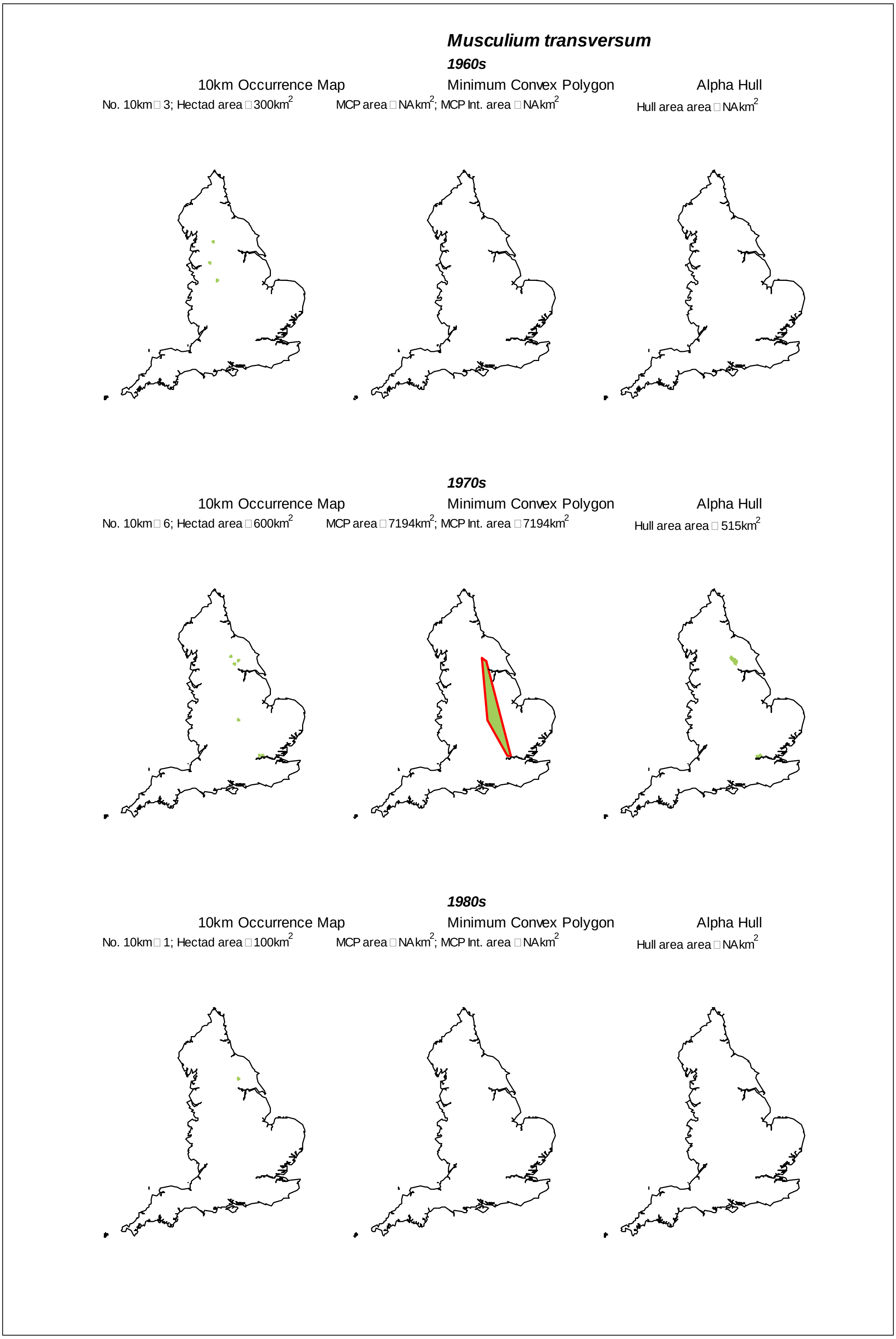


Figure C.53a Change in area of extent for *Musculium transversum* by decade (1960s to 1980s)

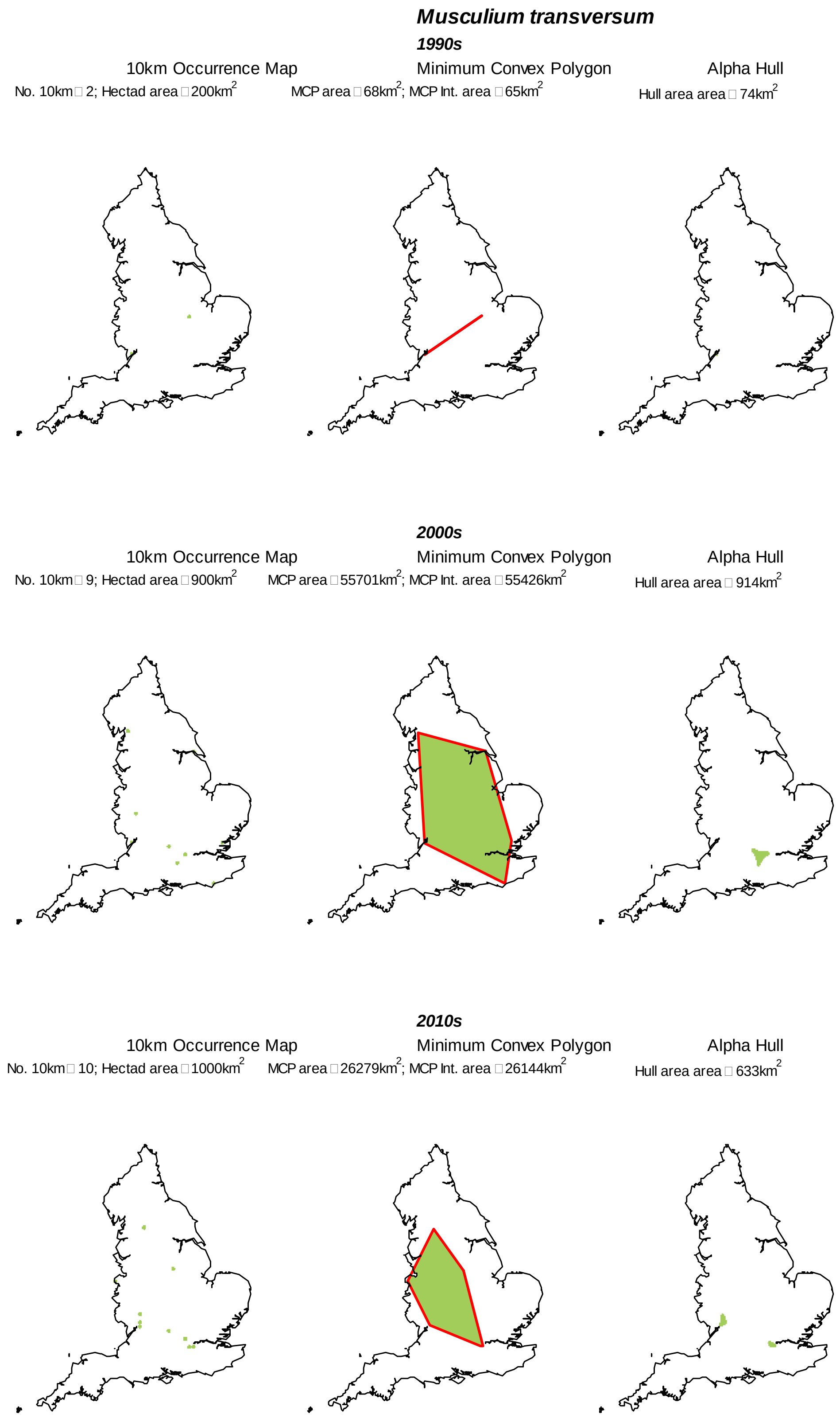


Figure C.53b Change in area of extent for *Musculium transversum* by decade (1990s to 2010s)

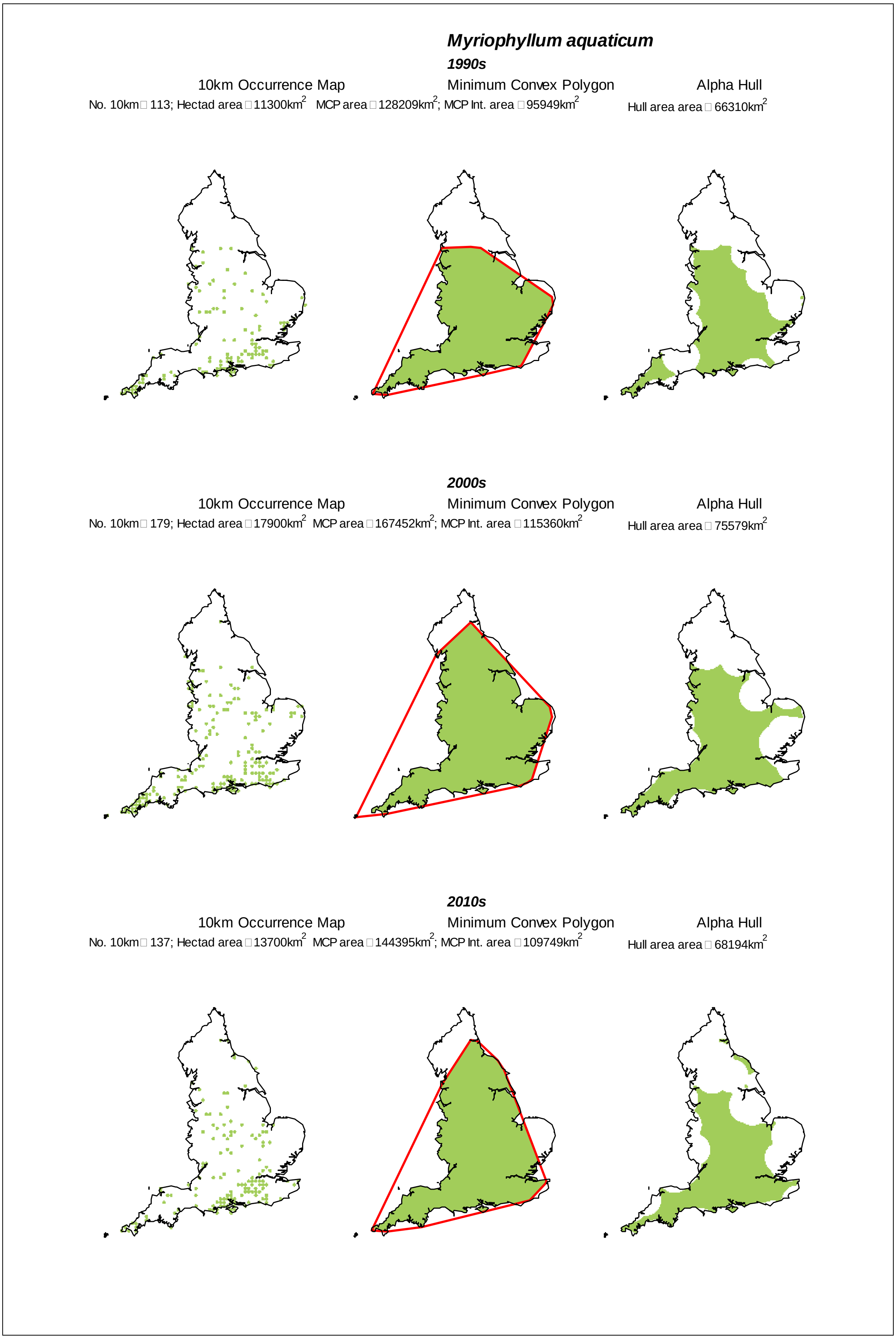


Figure C.54 Change in area of extent for *Myriophyllum aquaticum* by decade (1990s to 2010s)

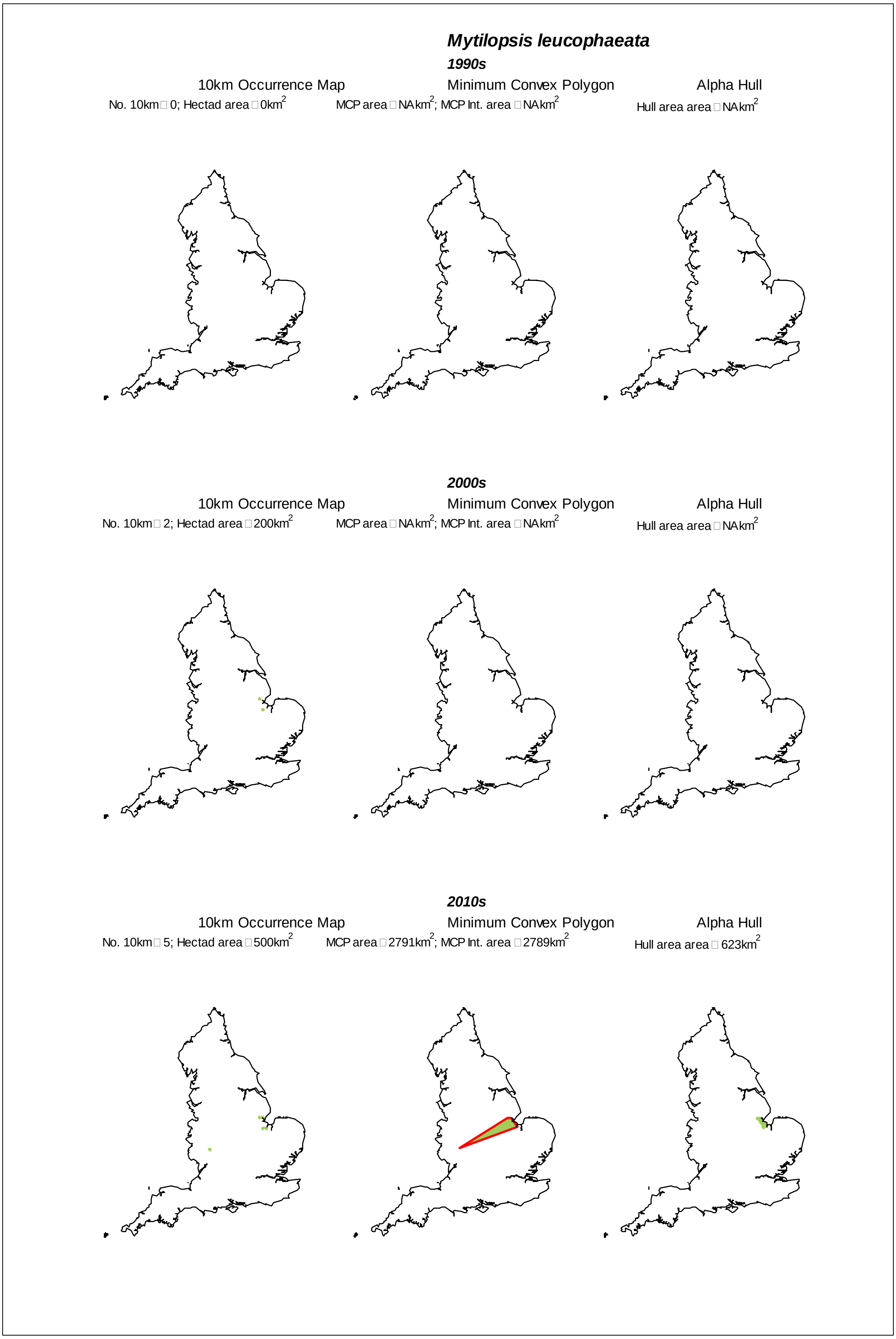


Figure C.55 Change in area of extent for *Mytilopsis leucophaeata* by decade (1990s to 2010s)

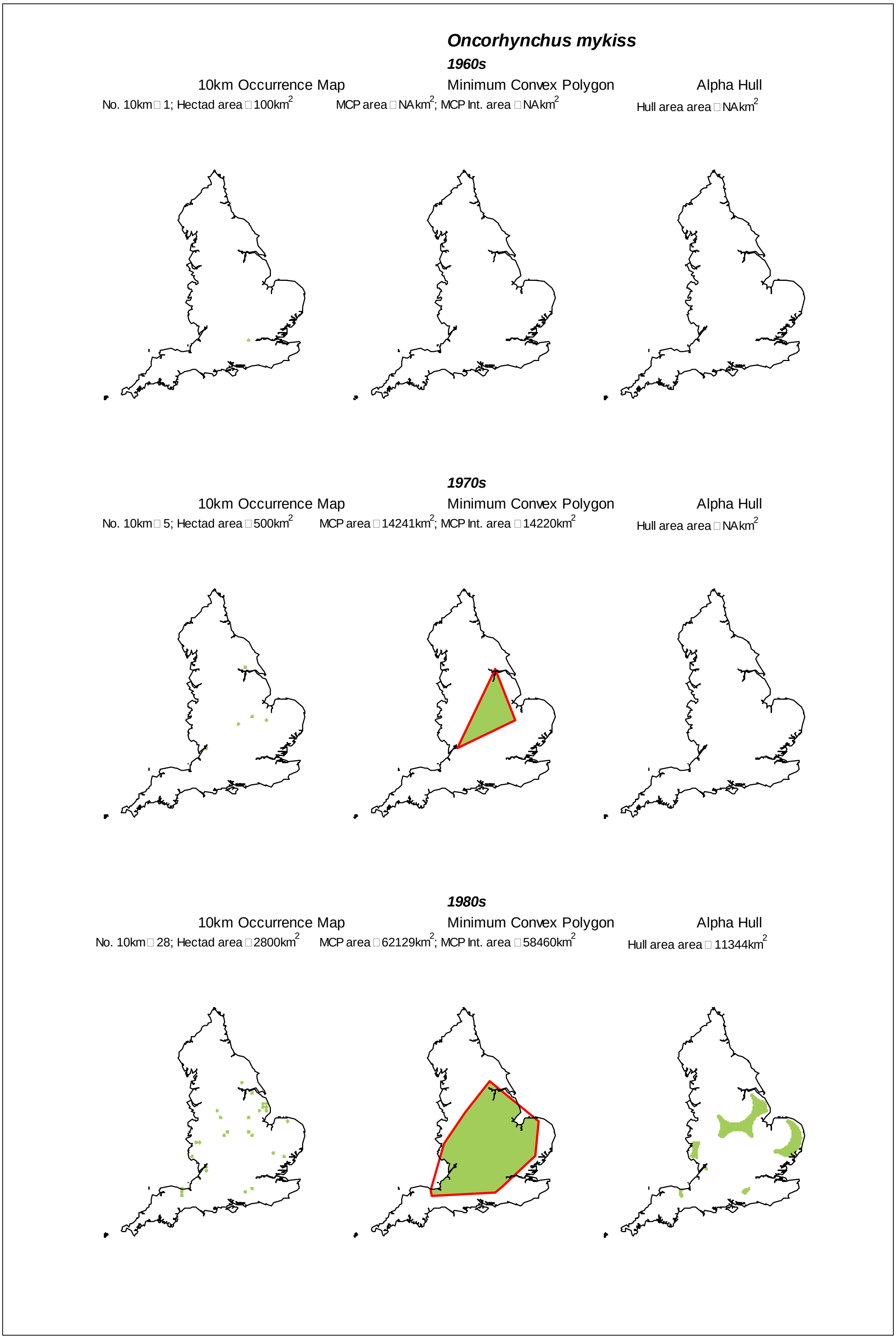


Figure C.56a Change in area of extent for *Oncorhynchus mykiss* by decade (1960s to 1980s)

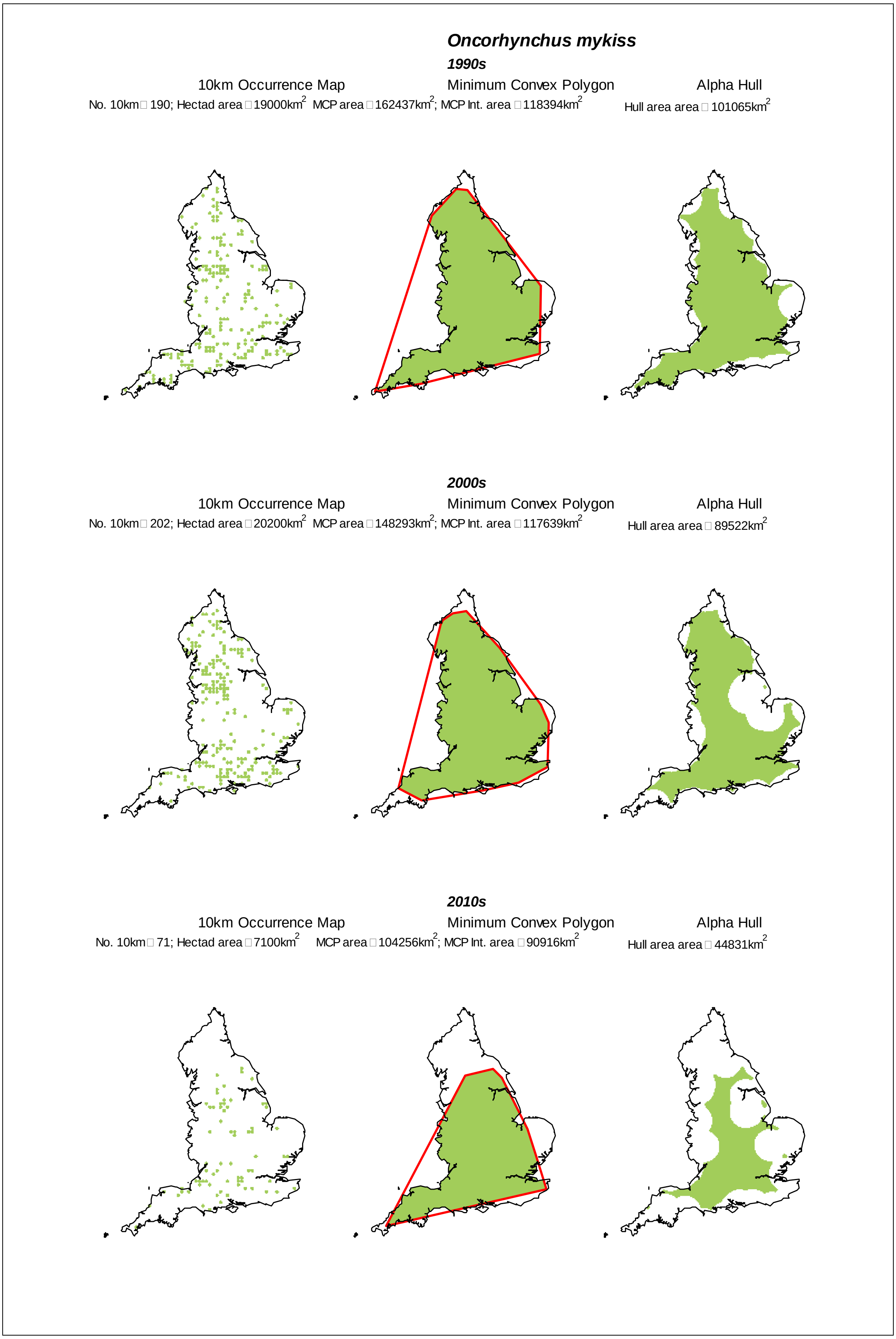


Figure C.56b Change in area of extent for *Oncorhynchus mykiss* by decade (1990s to 2010s)

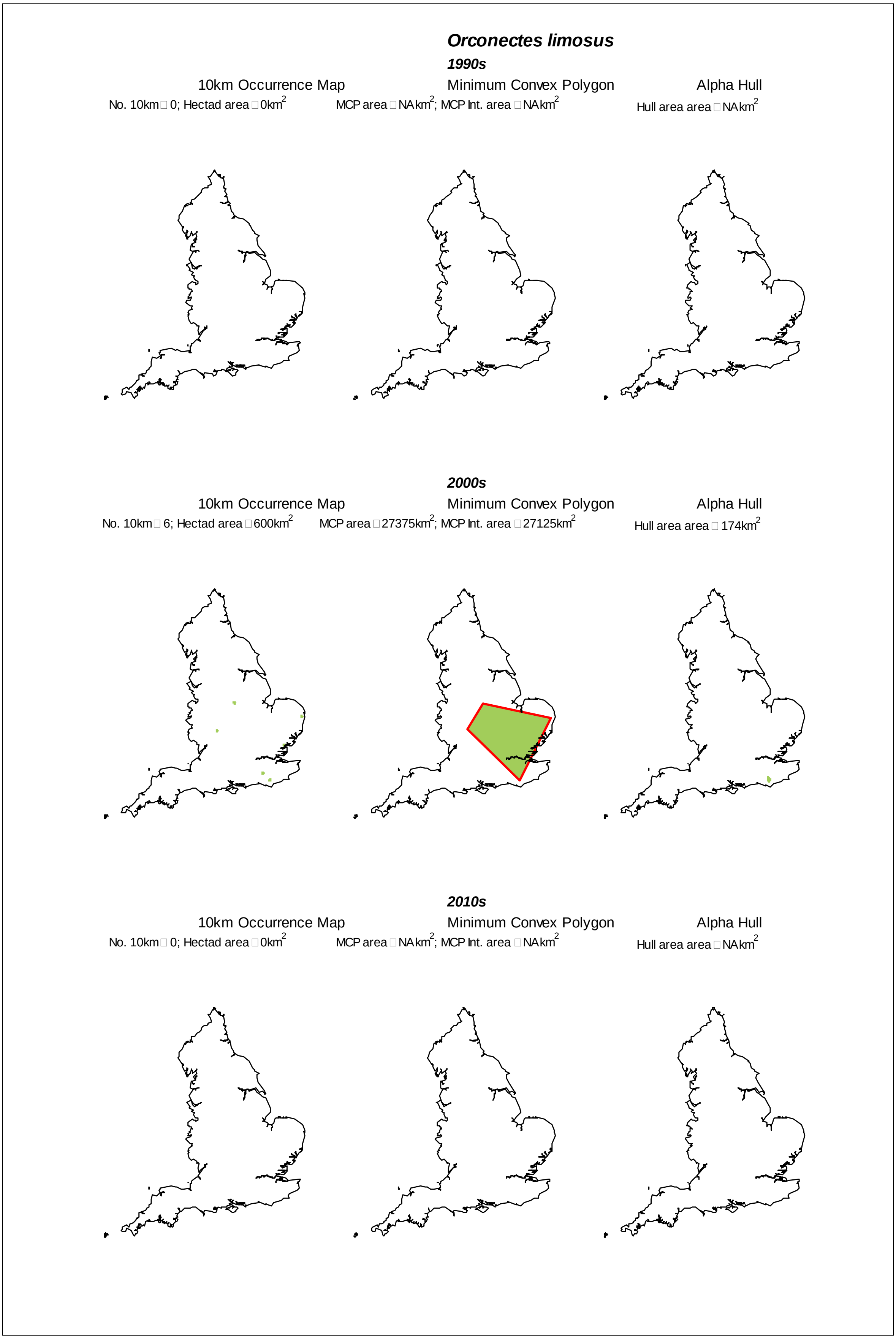


Figure C.57 Change in area of extent for *Orconectes limosus* by decade (1990s to 2010s)

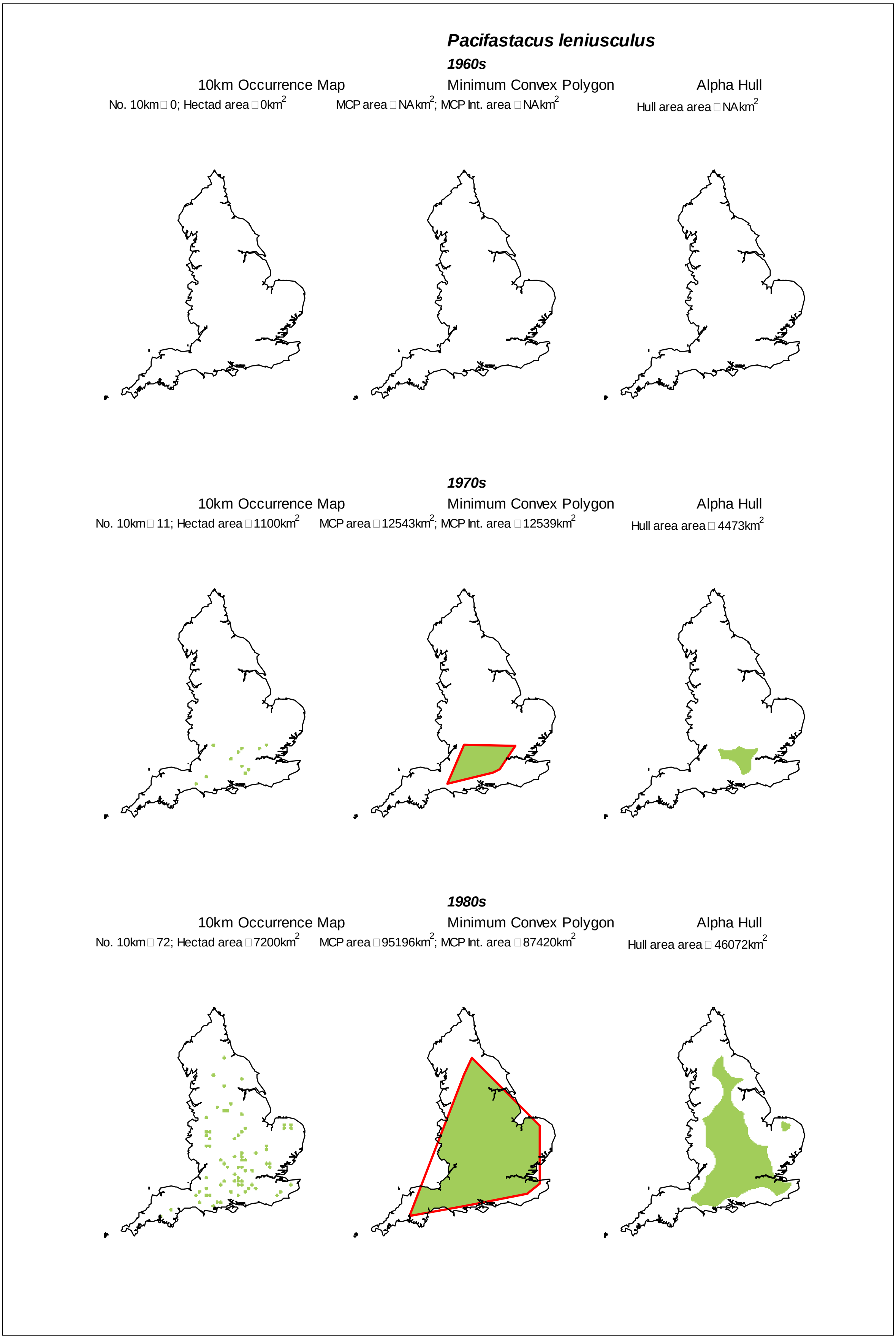
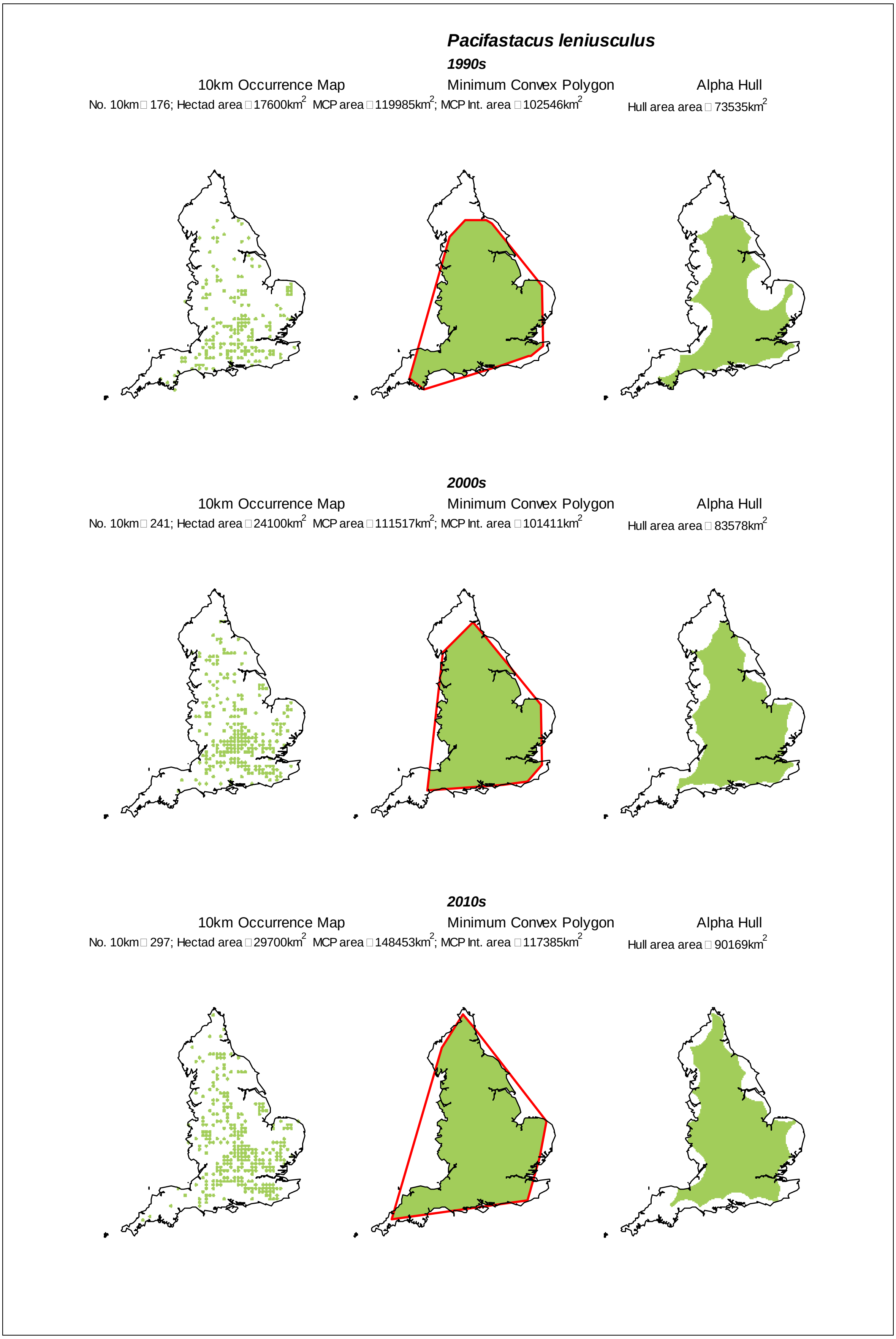


Figure C.58a Change in area of extent for *Pacifastacus leniusculus* by decade (1960s to 1980s)



Pacifastacus leniusculus

1990s

10km Occurrence Map — No. 10km□ 176; Hectad area □17600km^2

Minimum Convex Polygon — MCP area □119985km^2; MCP Int. area □102546km^2

Alpha Hull — Hull area area □ 73535km^2

2000s

10km Occurrence Map — No. 10km□ 241; Hectad area □24100km^2

Minimum Convex Polygon — MCP area □111517km^2; MCP Int. area □101411km^2

Alpha Hull — Hull area area □ 83578km^2

2010s

10km Occurrence Map — No. 10km□ 297; Hectad area □29700km^2

Minimum Convex Polygon — MCP area □148453km^2; MCP Int. area □117385km^2

Alpha Hull — Hull area area □ 90169km^2

Figure C.58b Change in area of extent for *Pacifastacus leniusculus* by decade (1990s to 2010s)

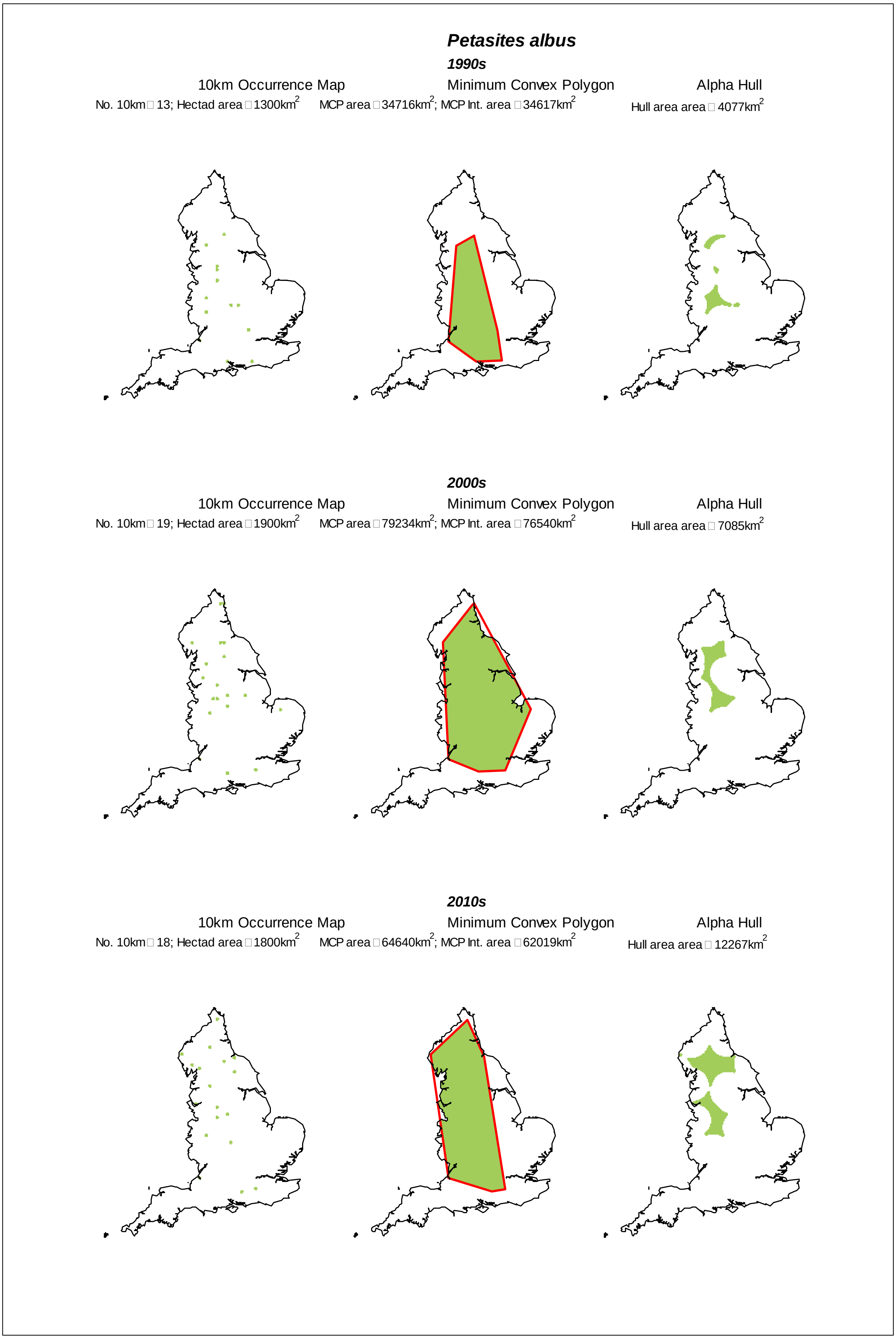


Figure C.59 Change in area of extent for *Petasites albus* by decade (1990s to 2010s)

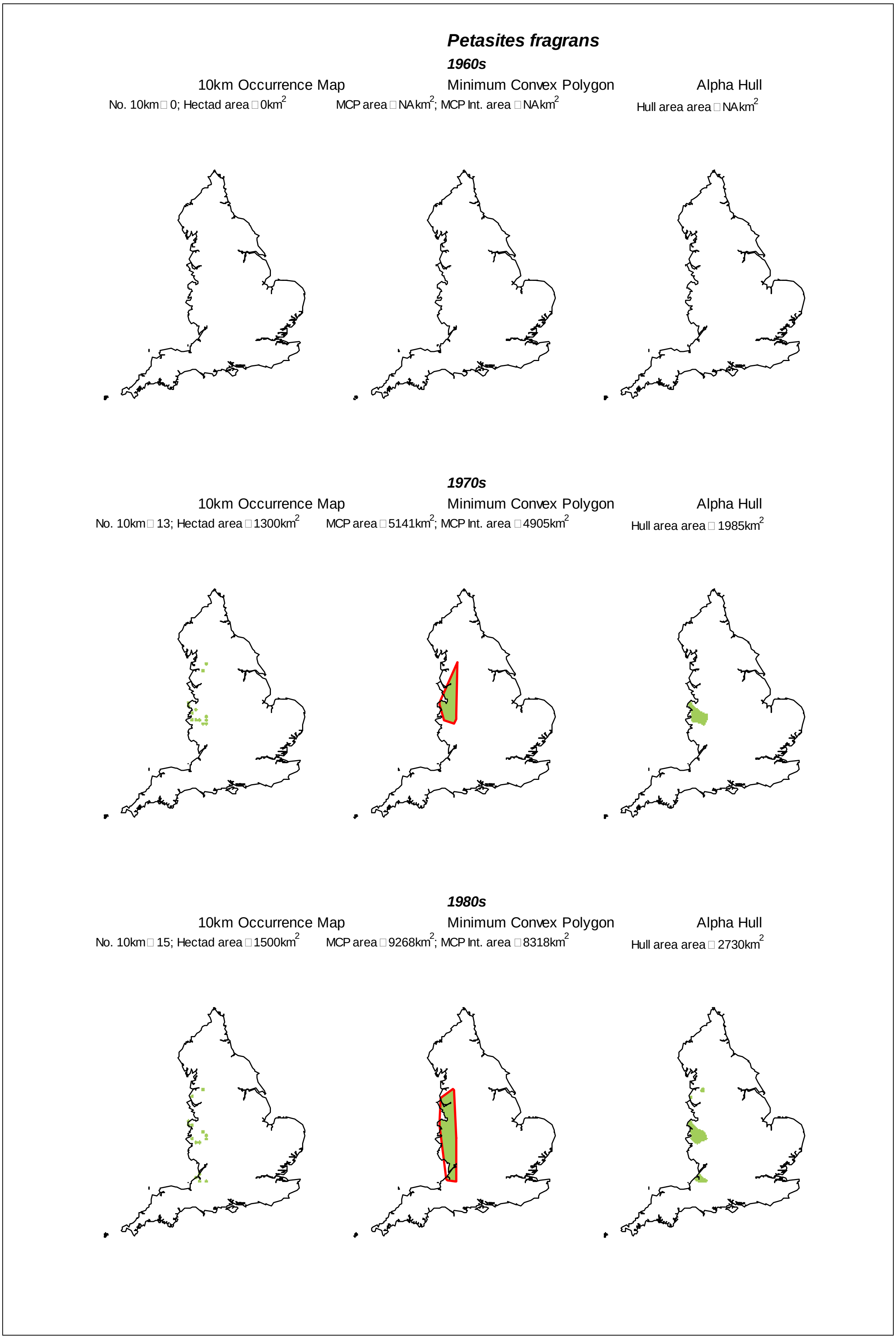


Figure C.60a Change in area of extent for *Petasites fragrans* by decade (1960s to 1980s)

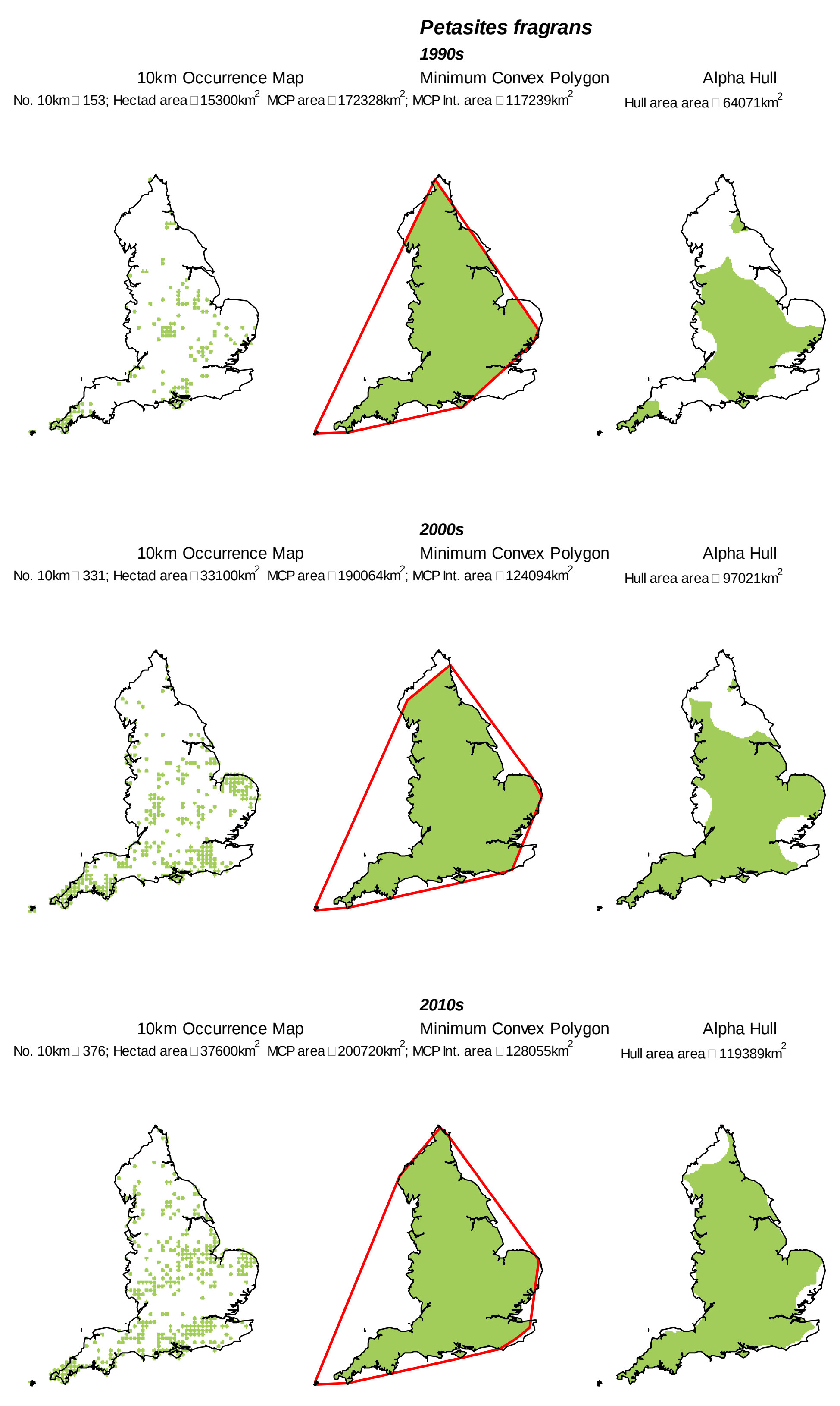


Figure C.60b Change in area of extent for *Petasites fragrans* by decade (1990s to 2010s)

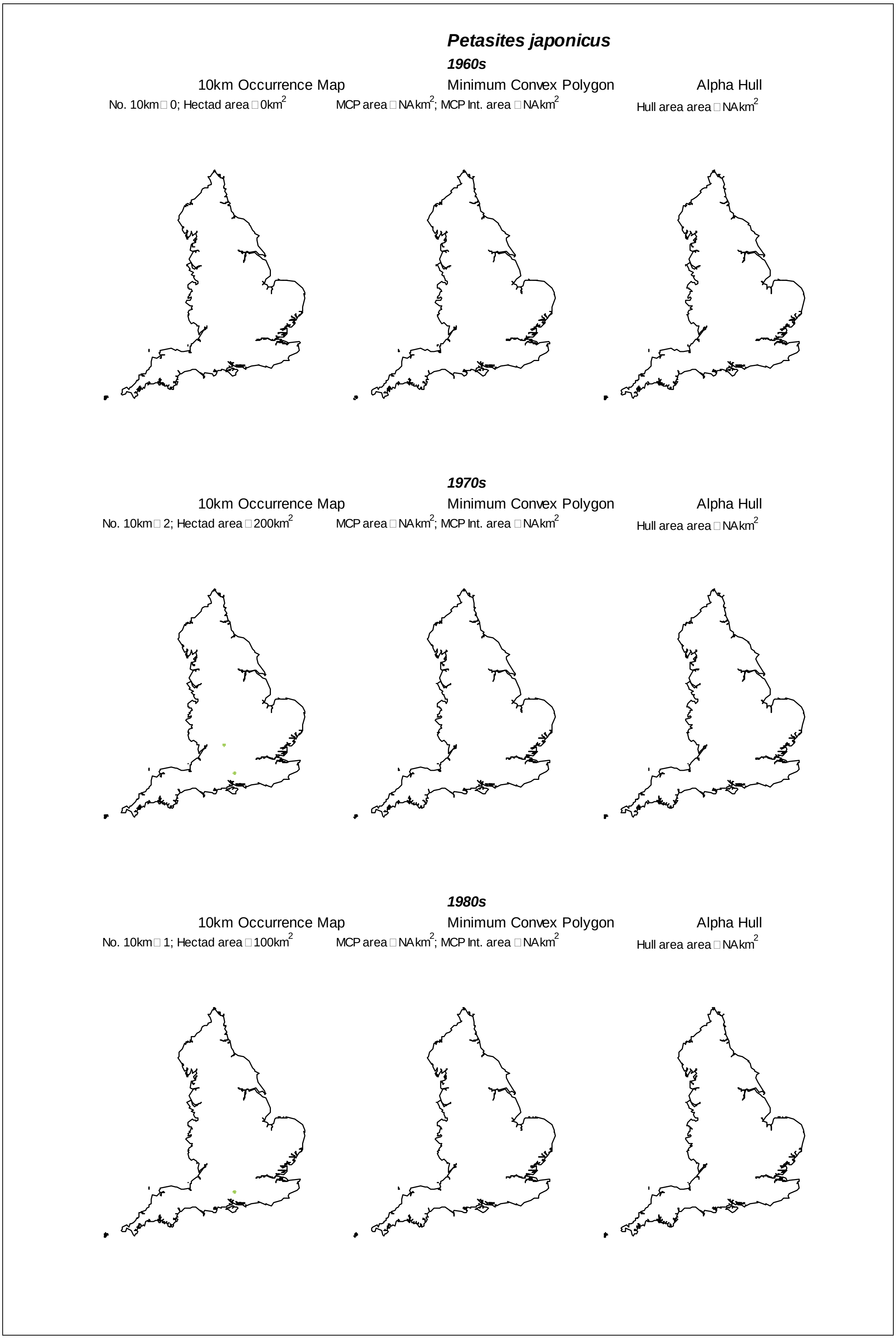


Figure C.61a Change in area of extent for *Petasites japonicus* by decade (1960s to 1980s)

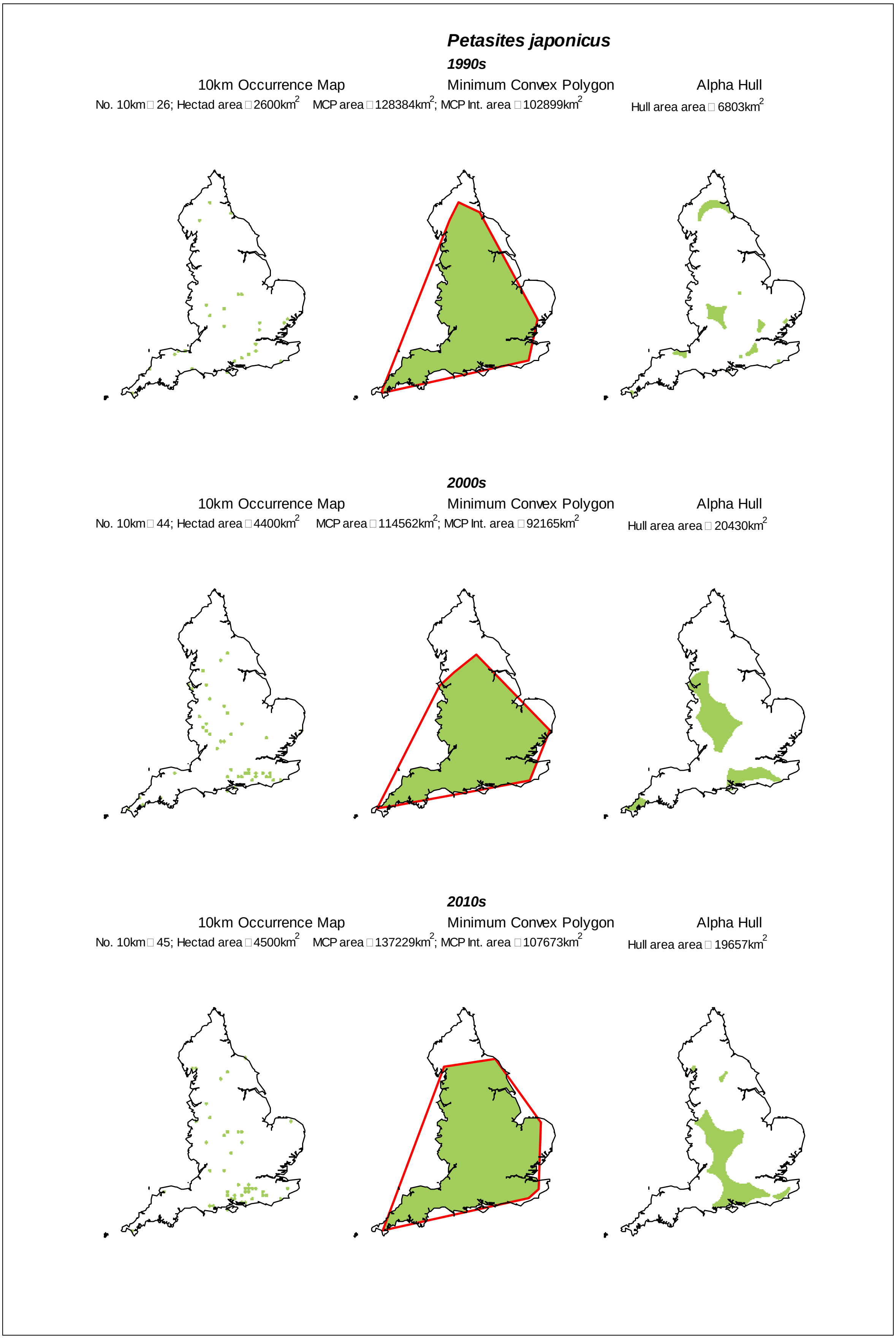


Figure C.61b Change in area of extent for *Petasites japonicus* by decade (1990s to 2010s)

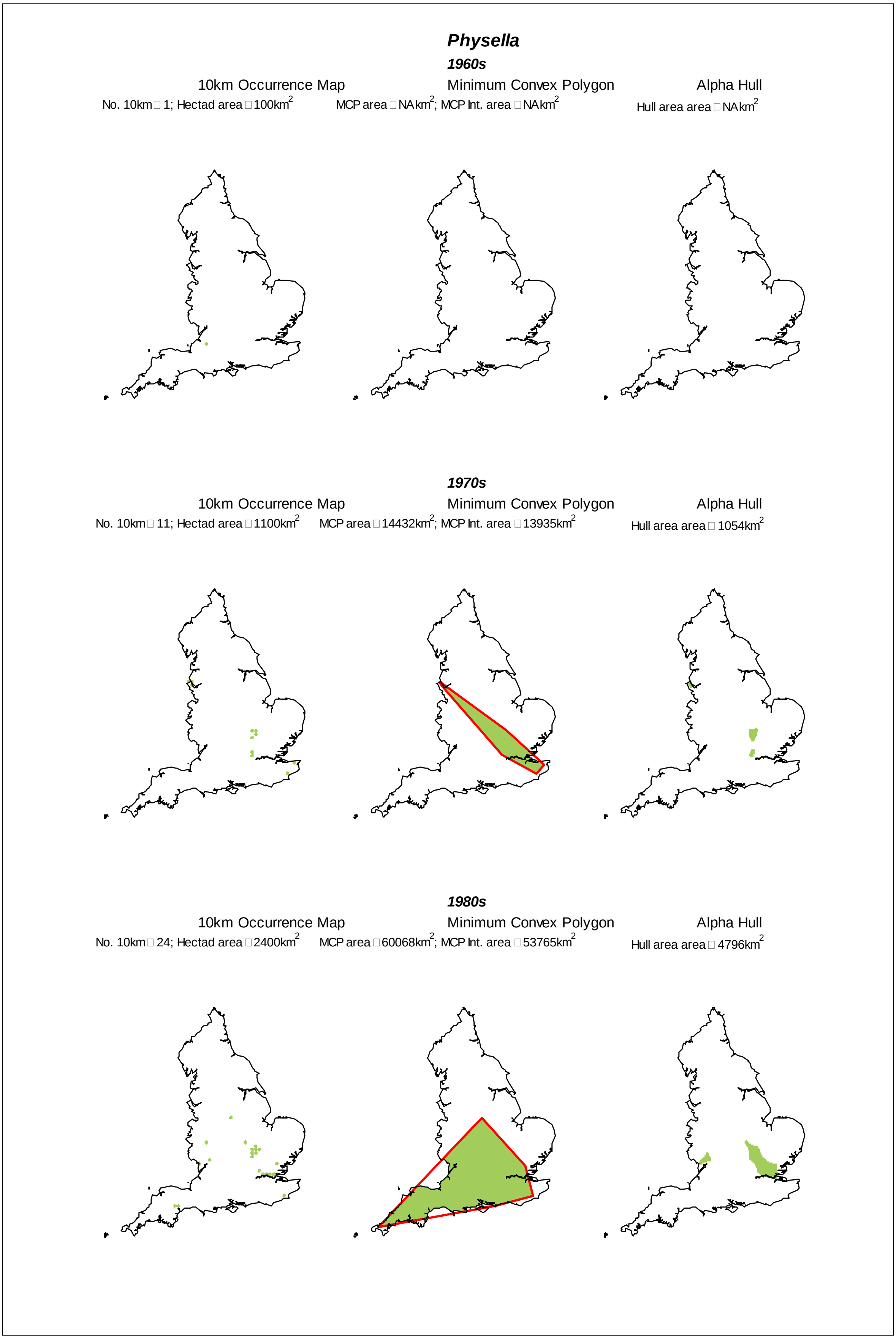


Figure C.62a Change in area of extent for *Physella* by decade (1960s to 1980s)

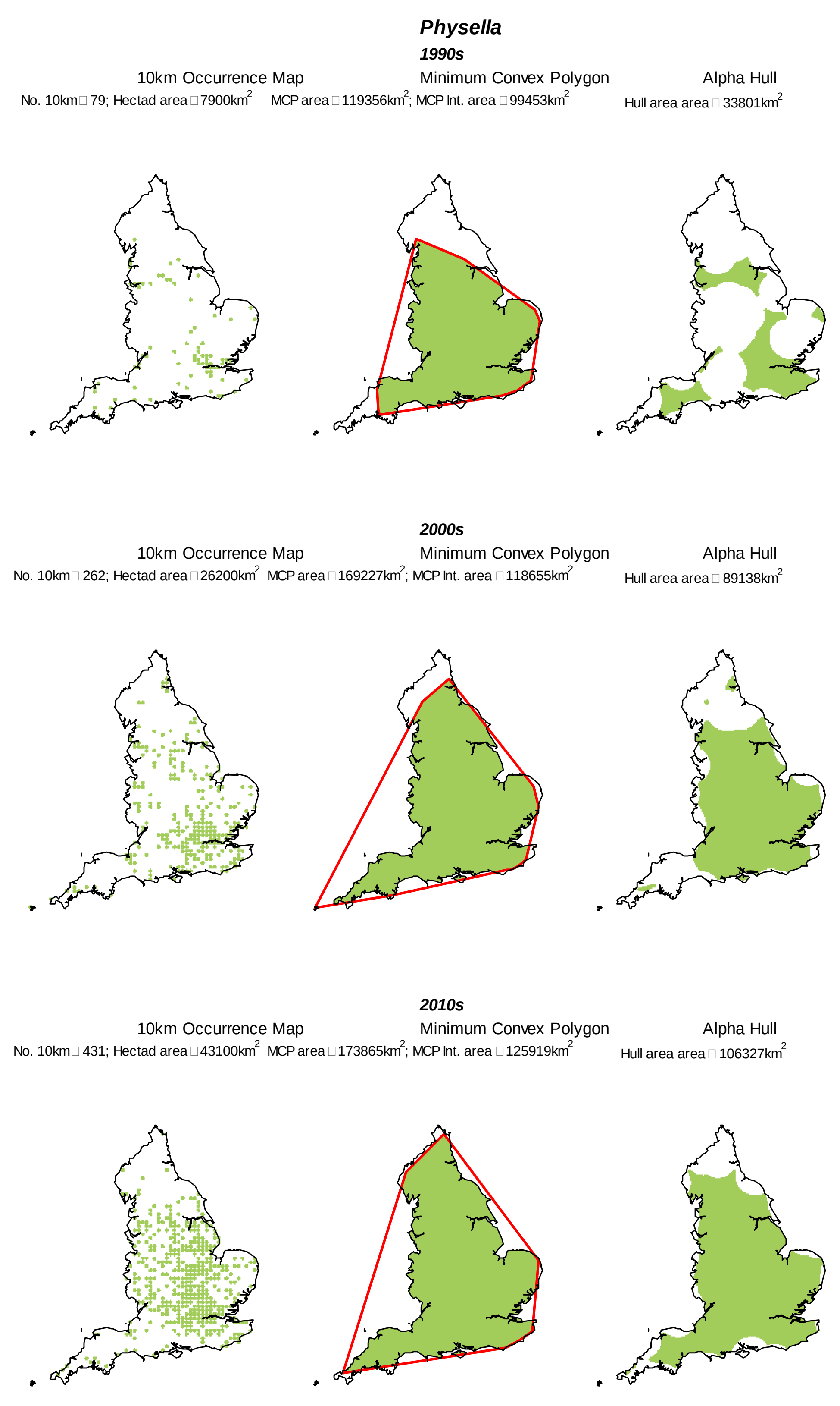


Figure C.62b Change in area of extent for *Physella* by decade (1990s to 2010s)

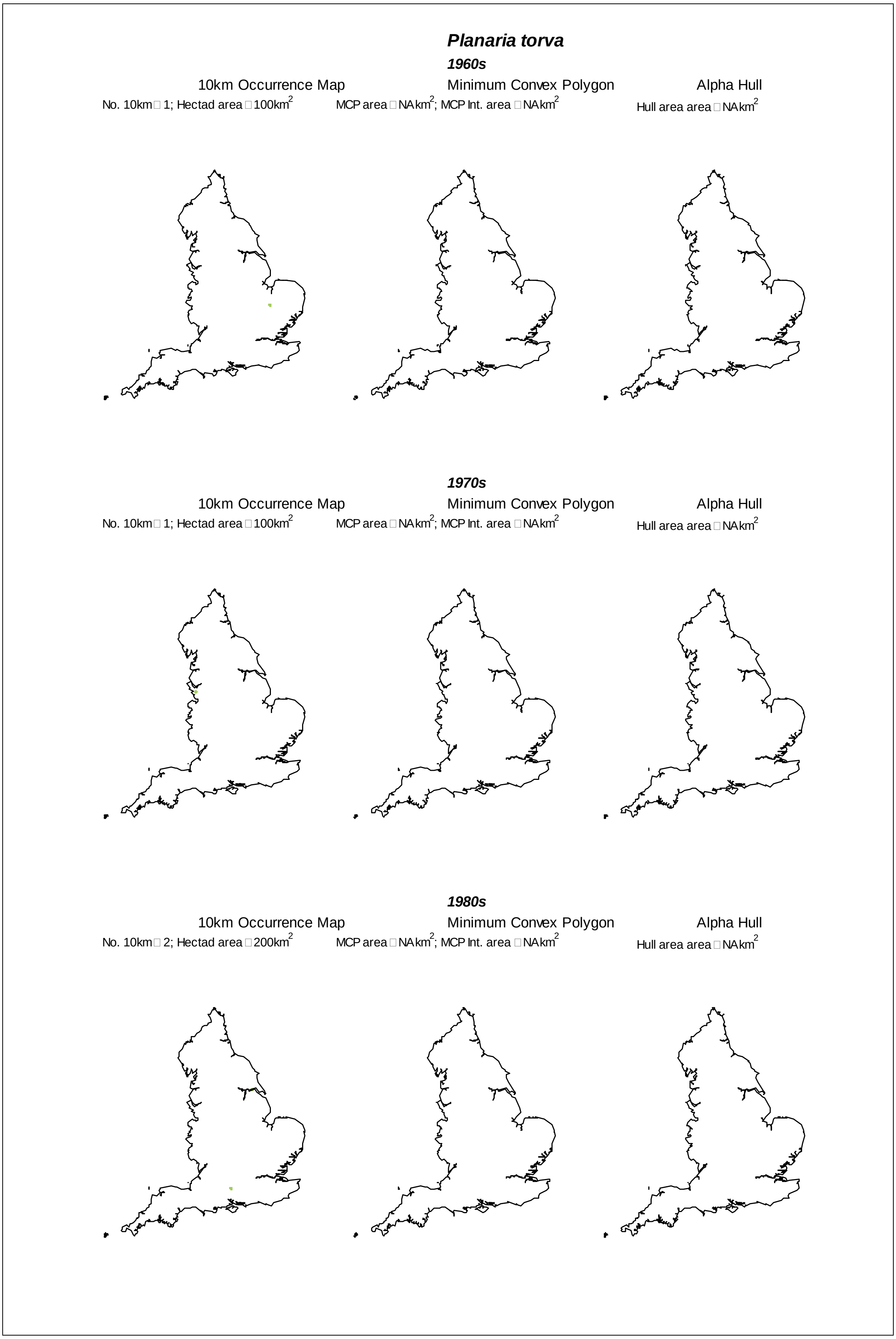


Figure C.63a Change in area of extent for *Planaria torva* by decade (1960s to 1980s)

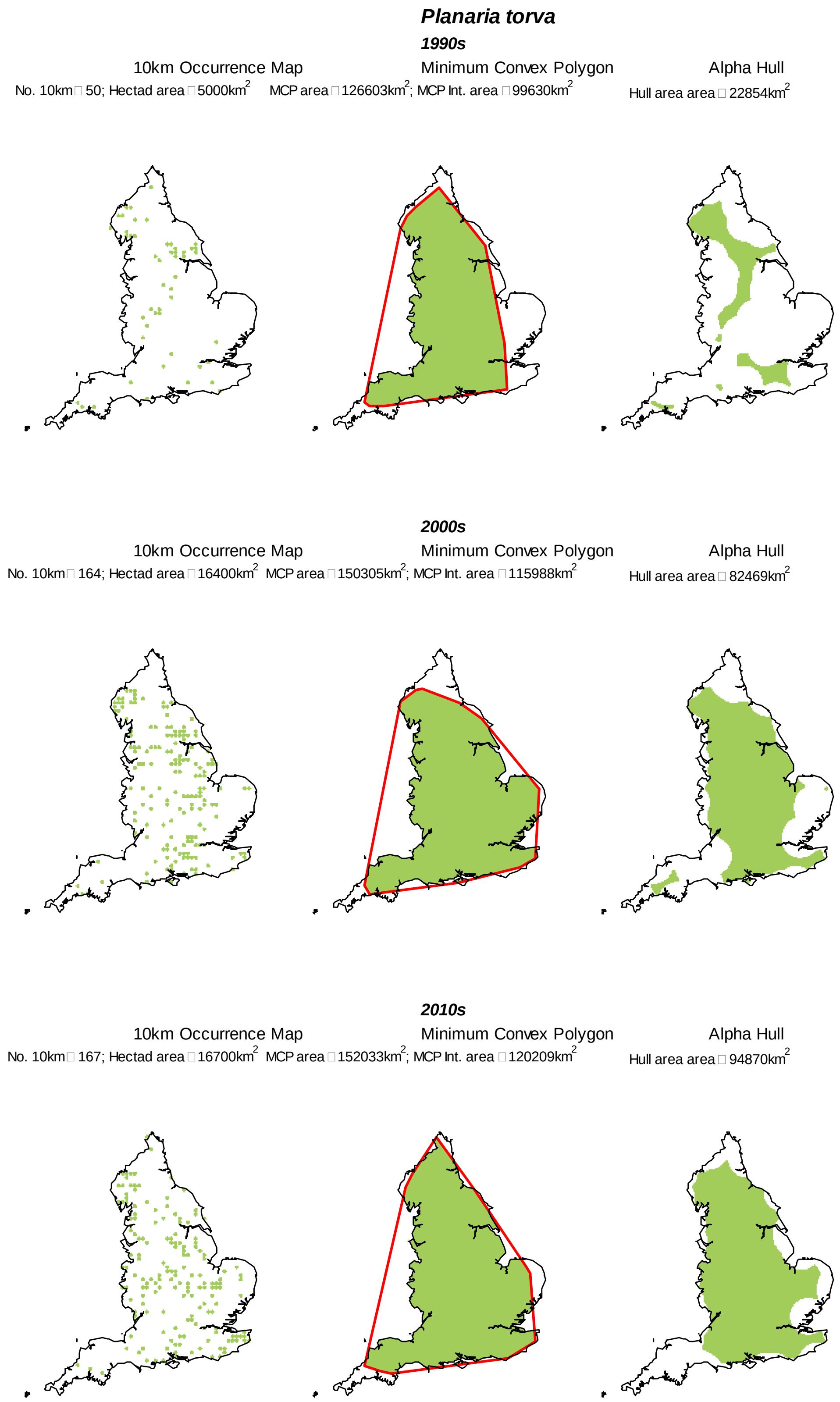


Figure C.63b Change in area of extent for *Planaria torva* by decade (1990s to 2010s)

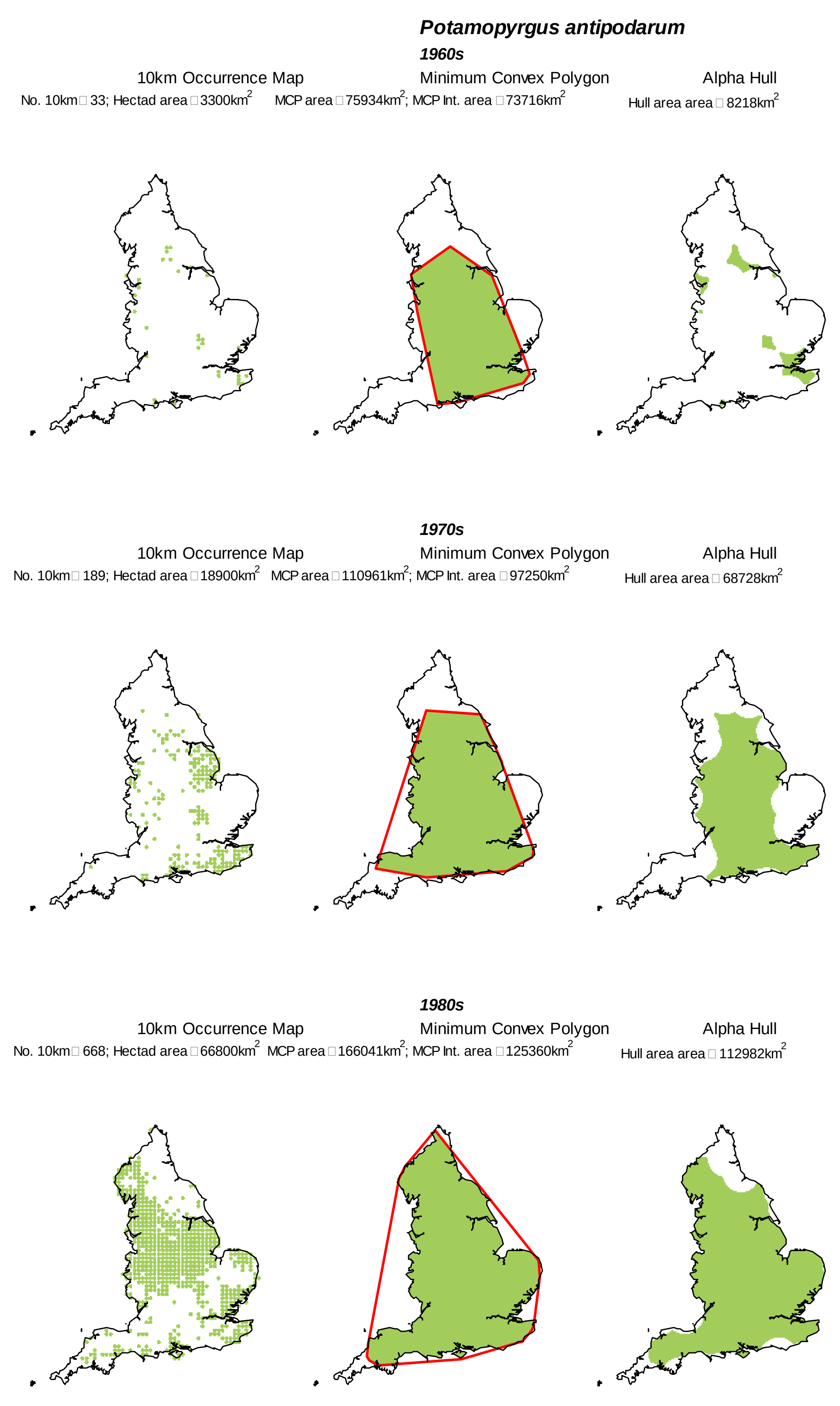


Figure C.64a Change in area of extent for *Potamopyrgus antipodarum* by decade (1960s to 1980s)

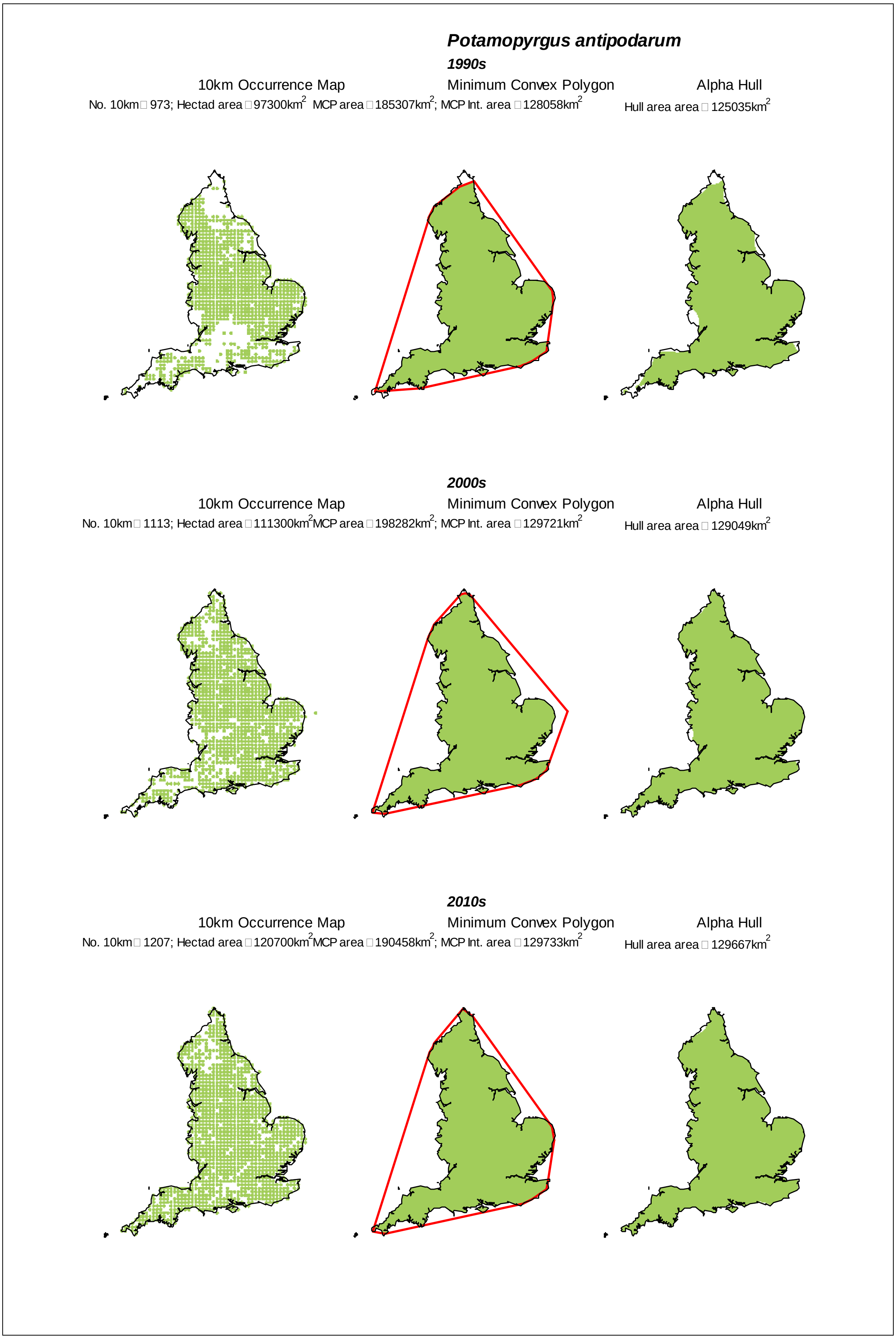


Figure C.64b Change in area of extent for *Potamopyrgus antipodarum* by decade (1990s to 2010s)

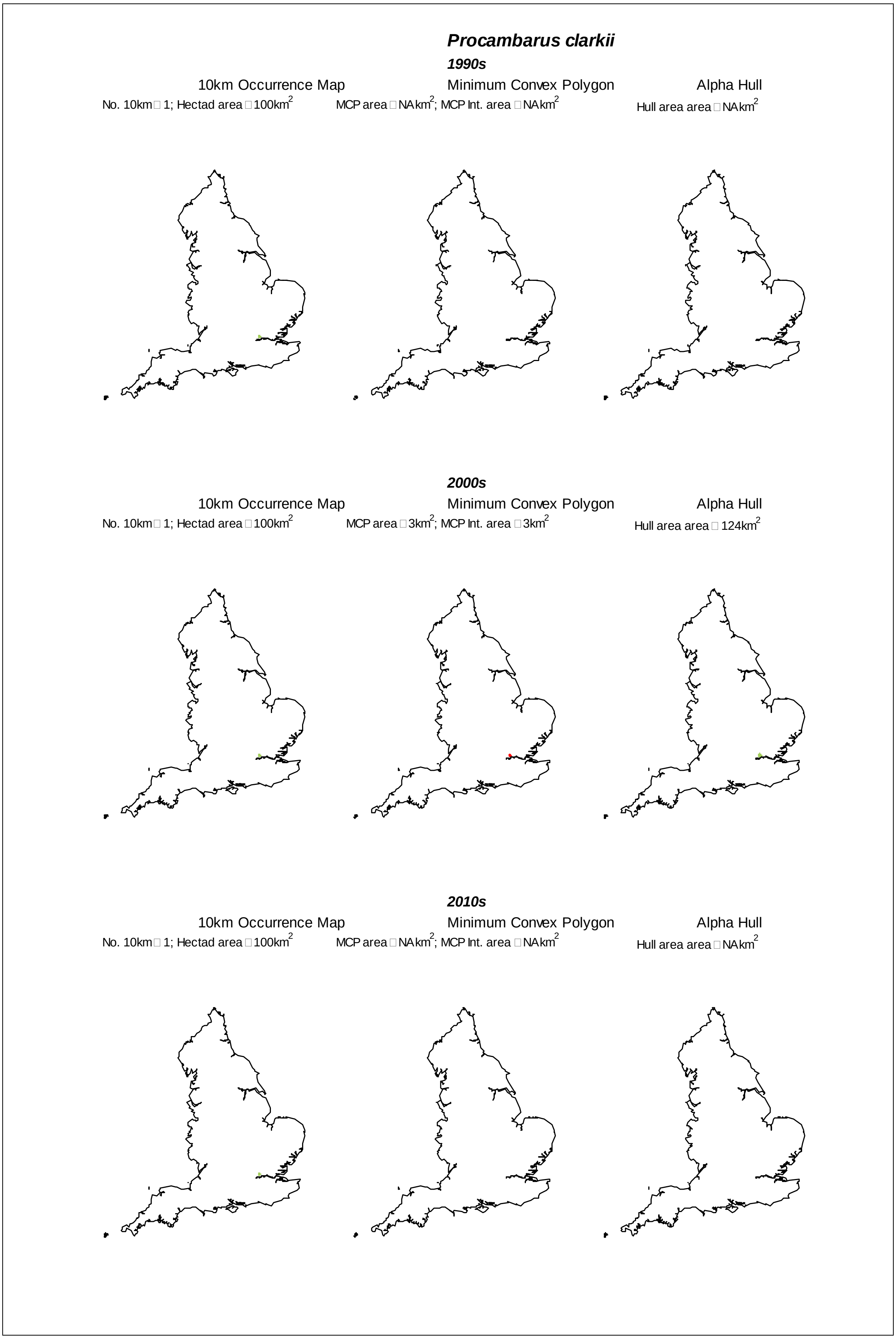


Figure C.65 Change in area of extent for *Procambarus clarkii* by decade

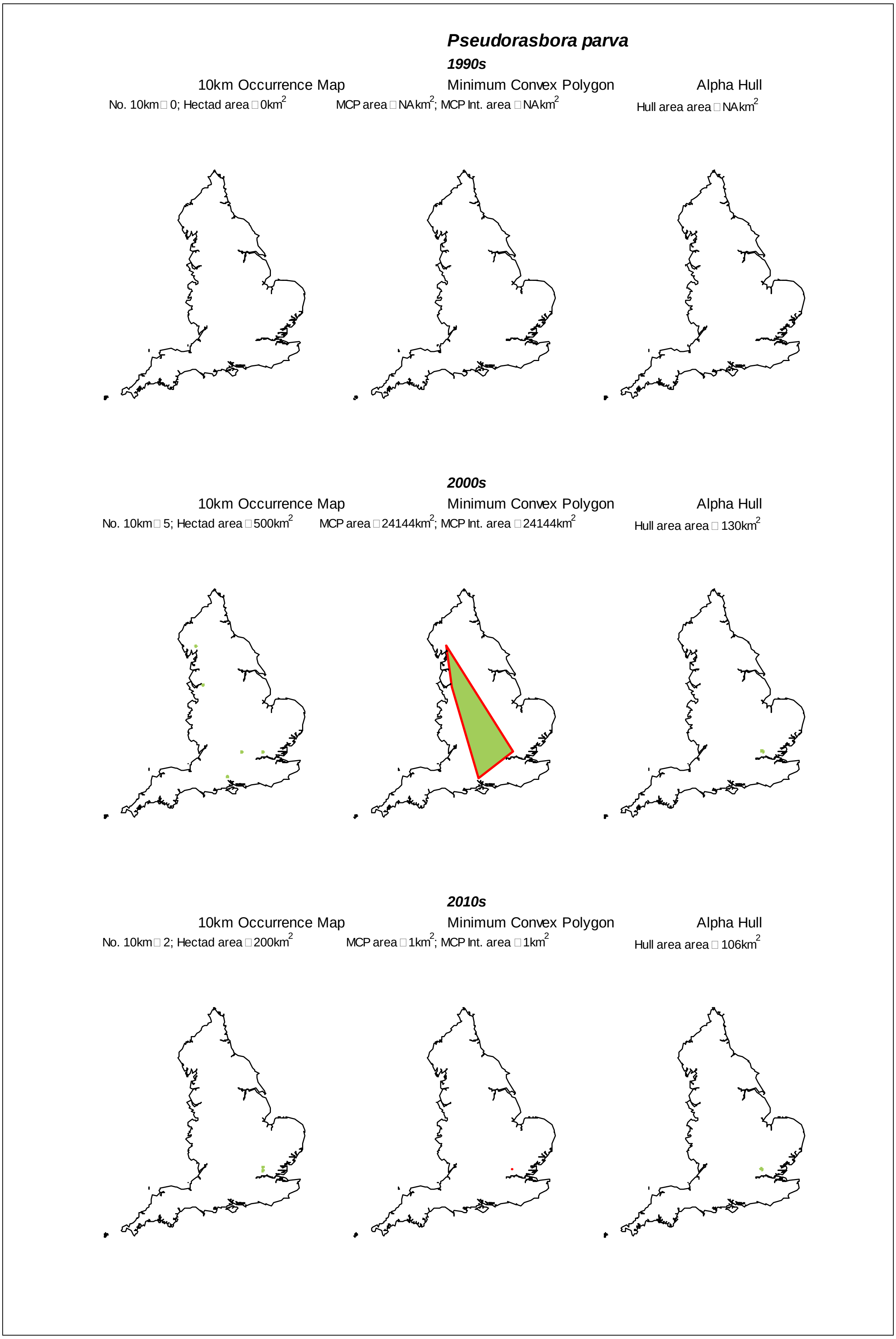


Figure C.66 Change in area of extent for *Pseudorasbora parva* by decade (1990s to 2010s)

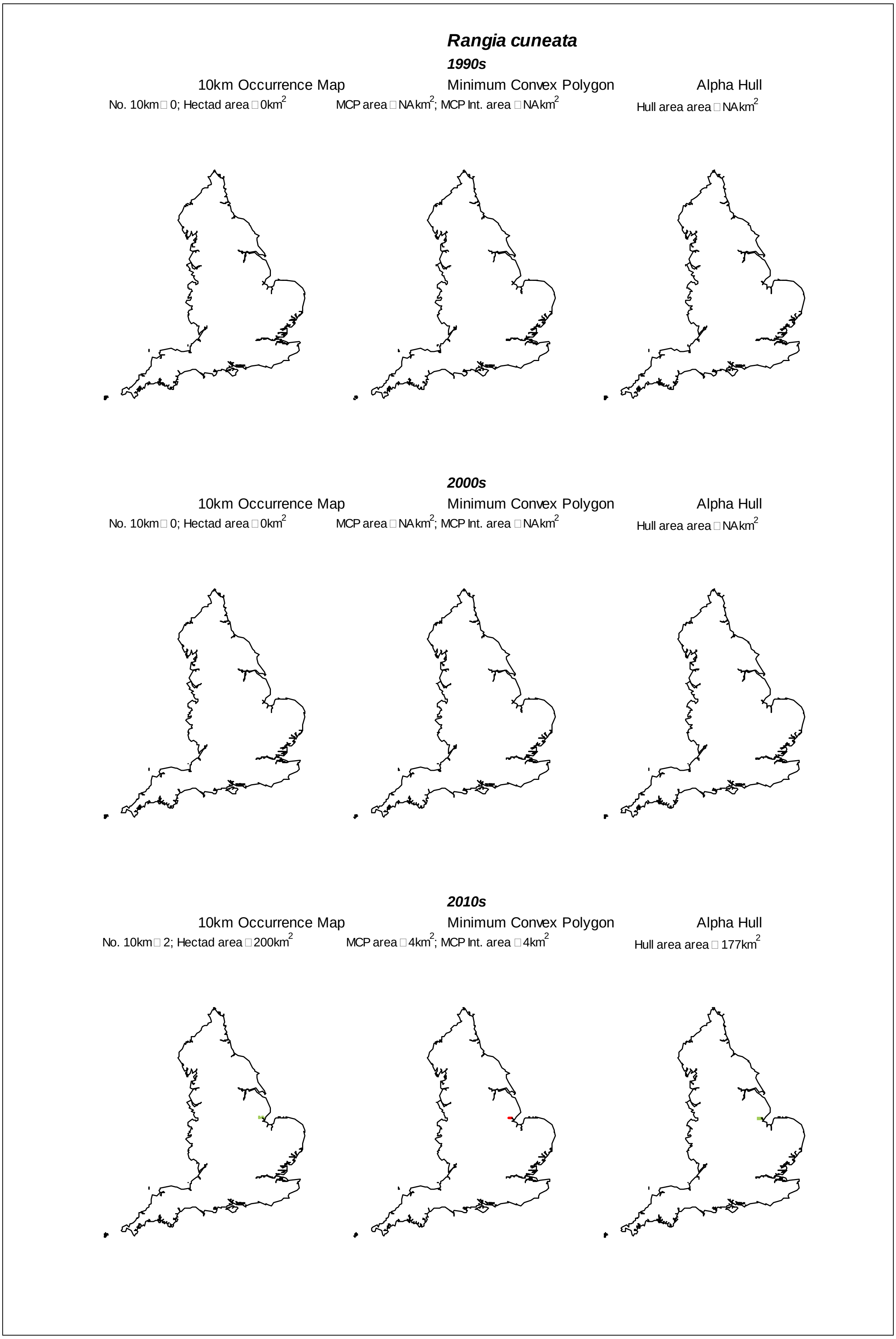


Figure C.67 Change in area of extent for *Rangia cuneate* by decade (1990s to 2010s)

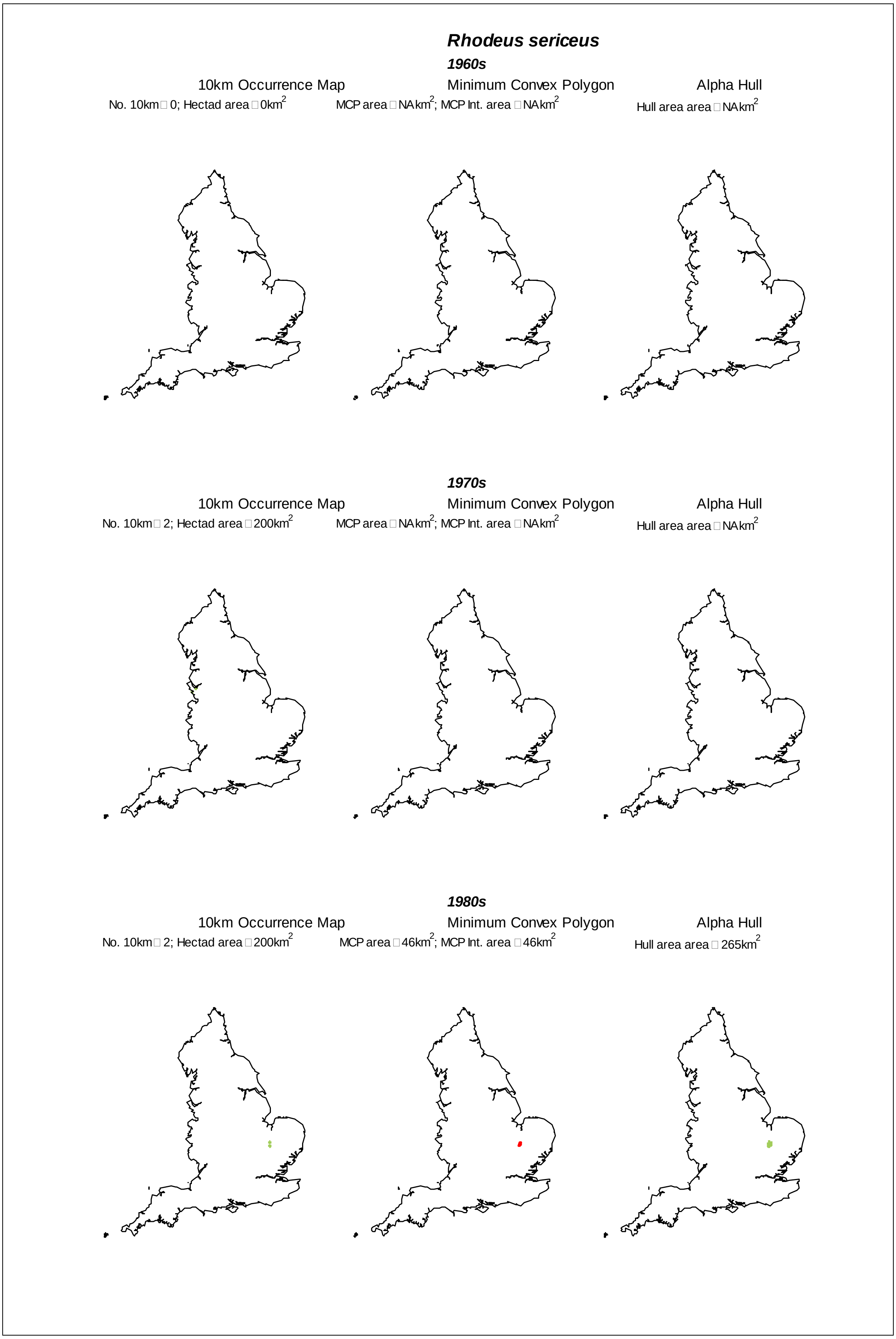


Figure C.68a Change in area of extent for *Rhodeus sericeus* by decade (1960s to 1980s)

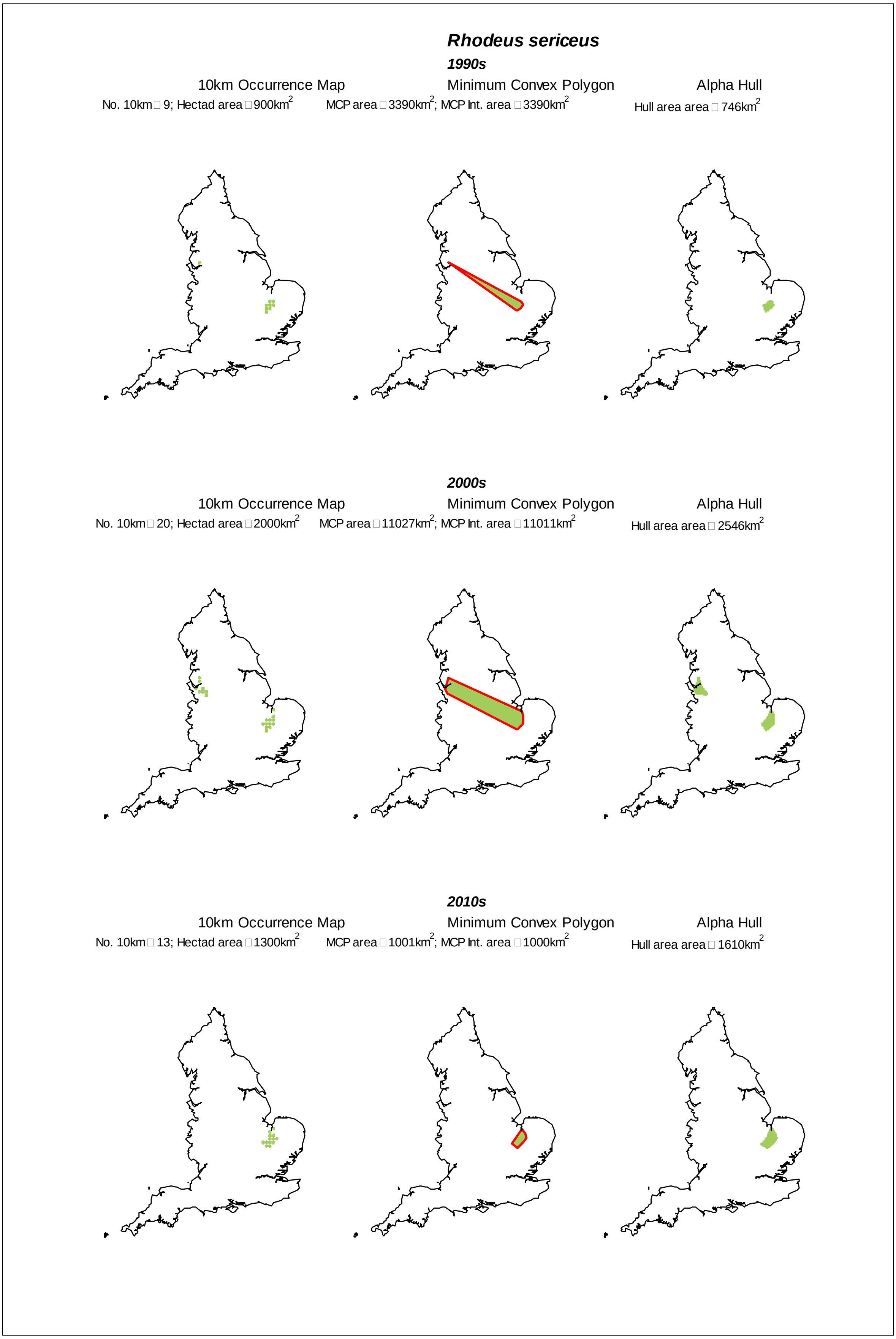


Figure C.68b Change in area of extent for *Rhodeus sericeus* by decade (1990s to 2010s)

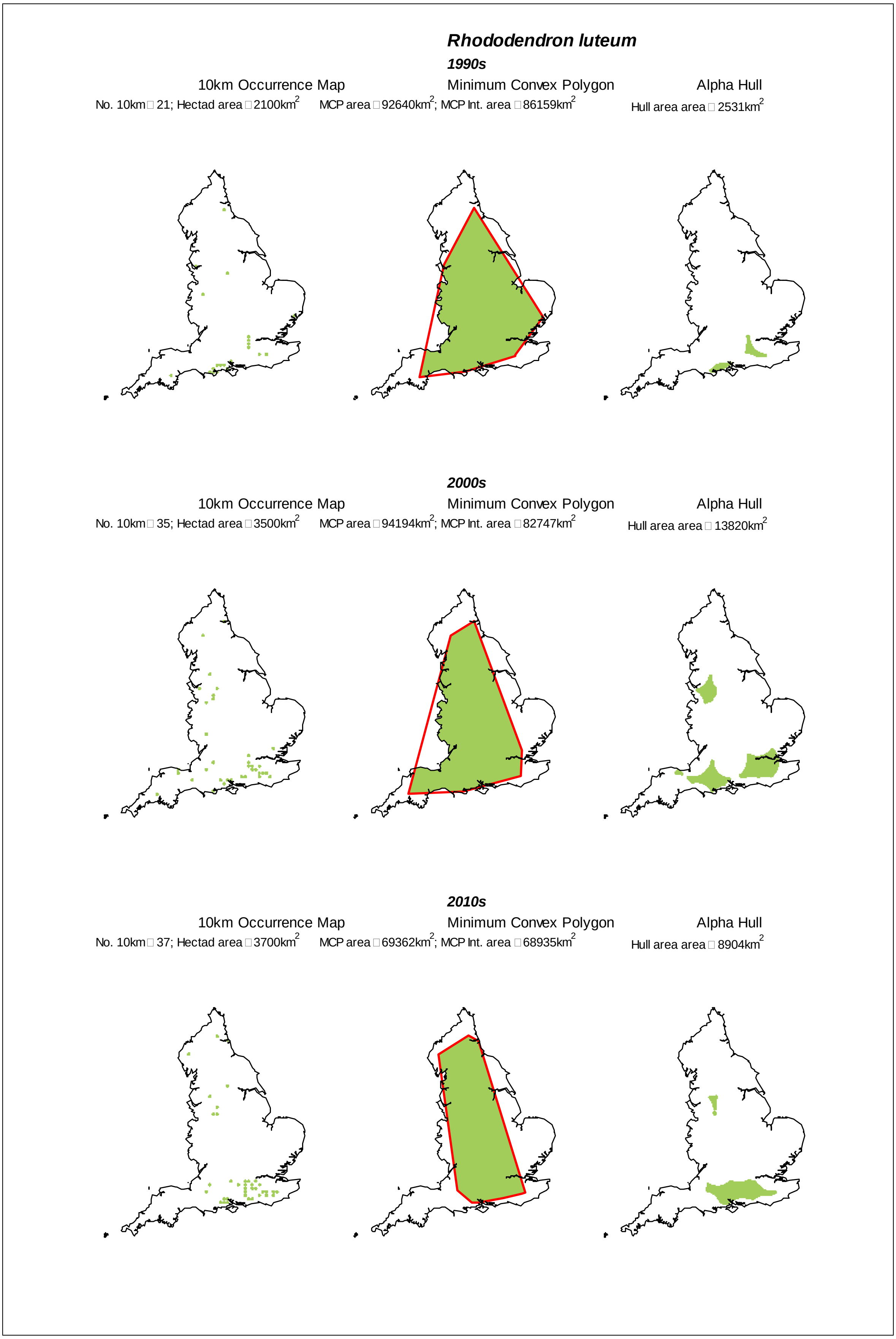


Figure C.69 Change in area of extent for *Rhododendron luteum* by decade (1990s to 2010s)

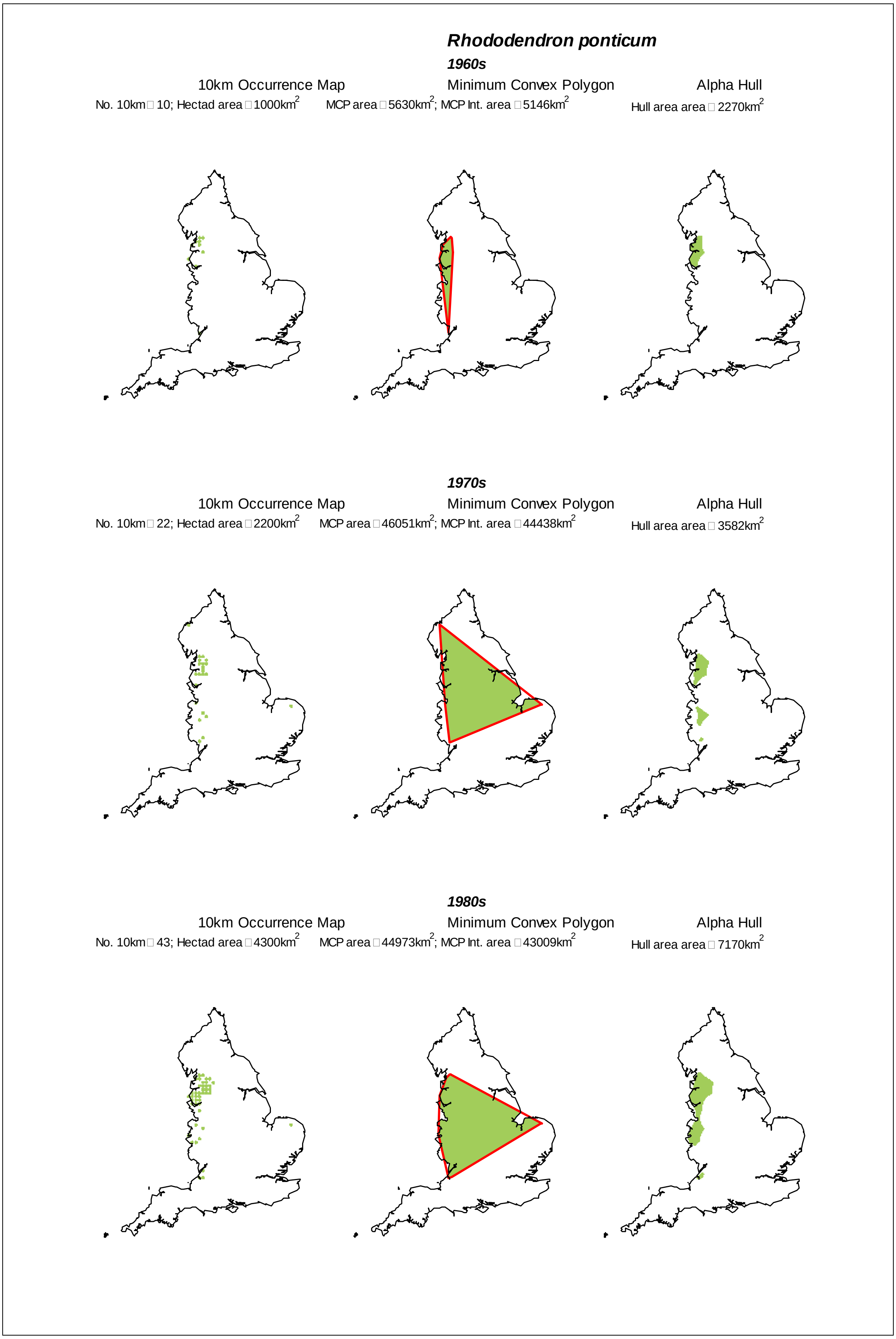


Figure C.70a Change in area of extent for *Rhododendron ponticum* by decade (1960s to 1980s)

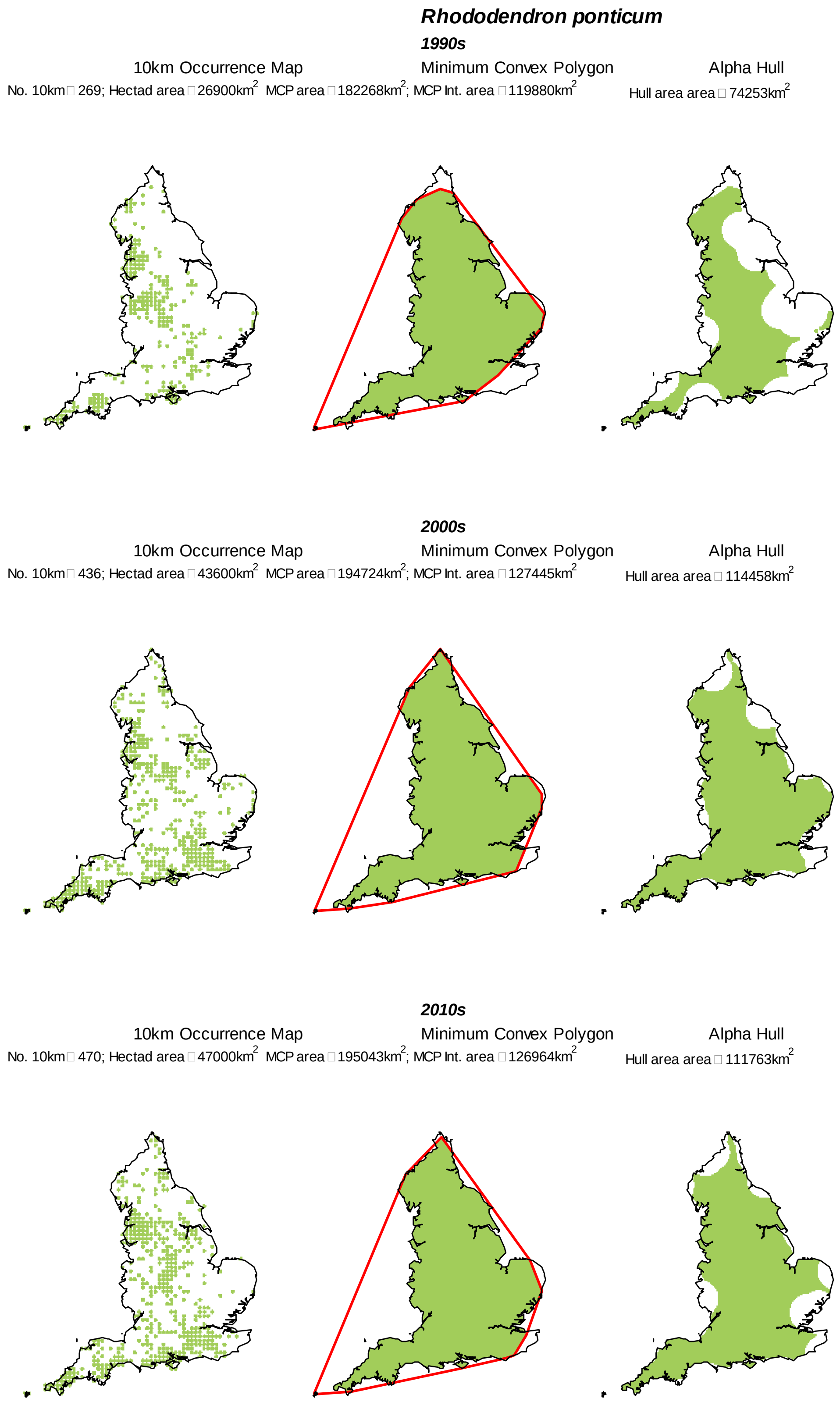


Figure C.70b Change in area of extent for *Rhododendron ponticum* by decade (1990s to 2010s)

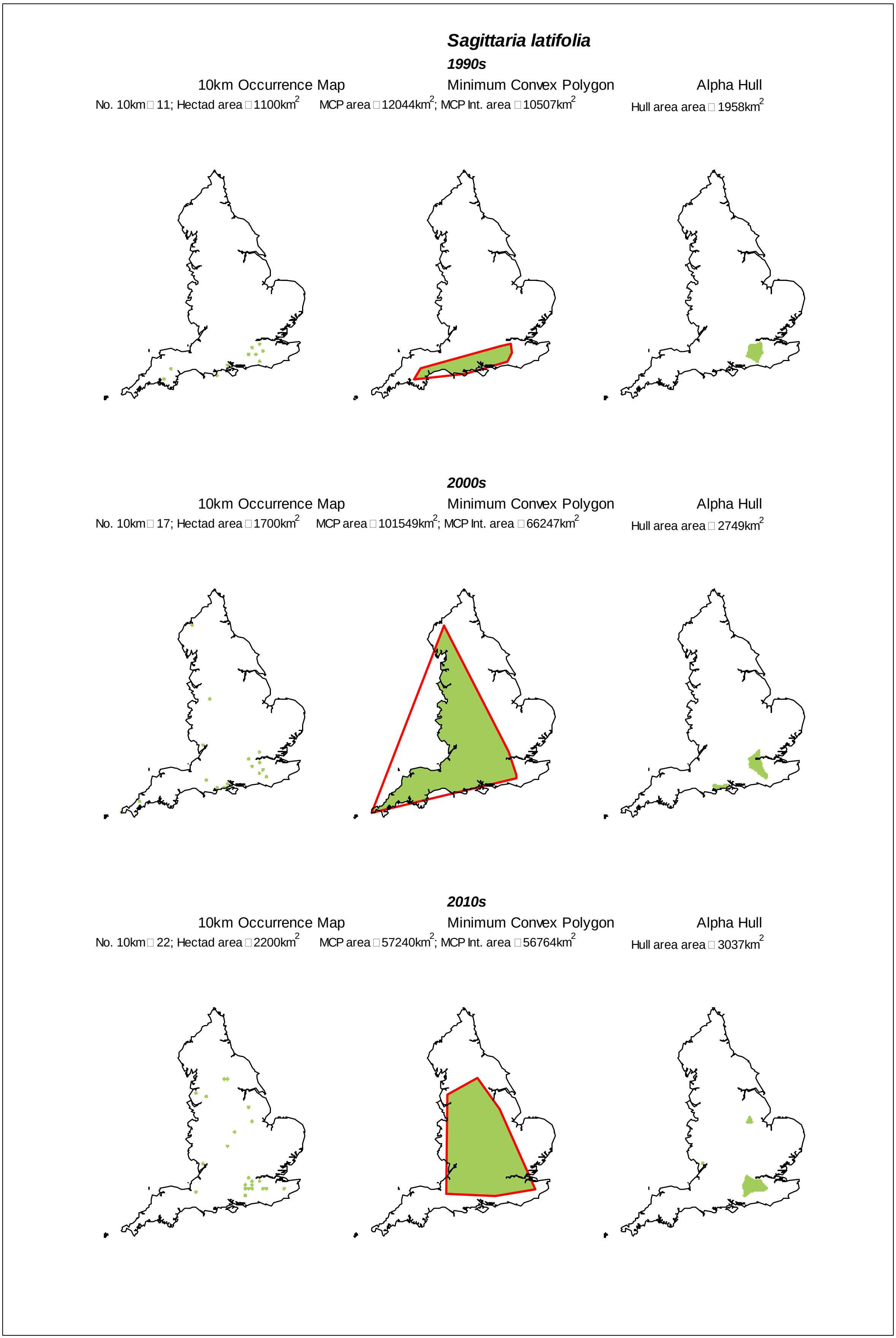


Figure C.71 Change in area of extent for *Sagittaria latifolia* by decade (1990s to 2010s)

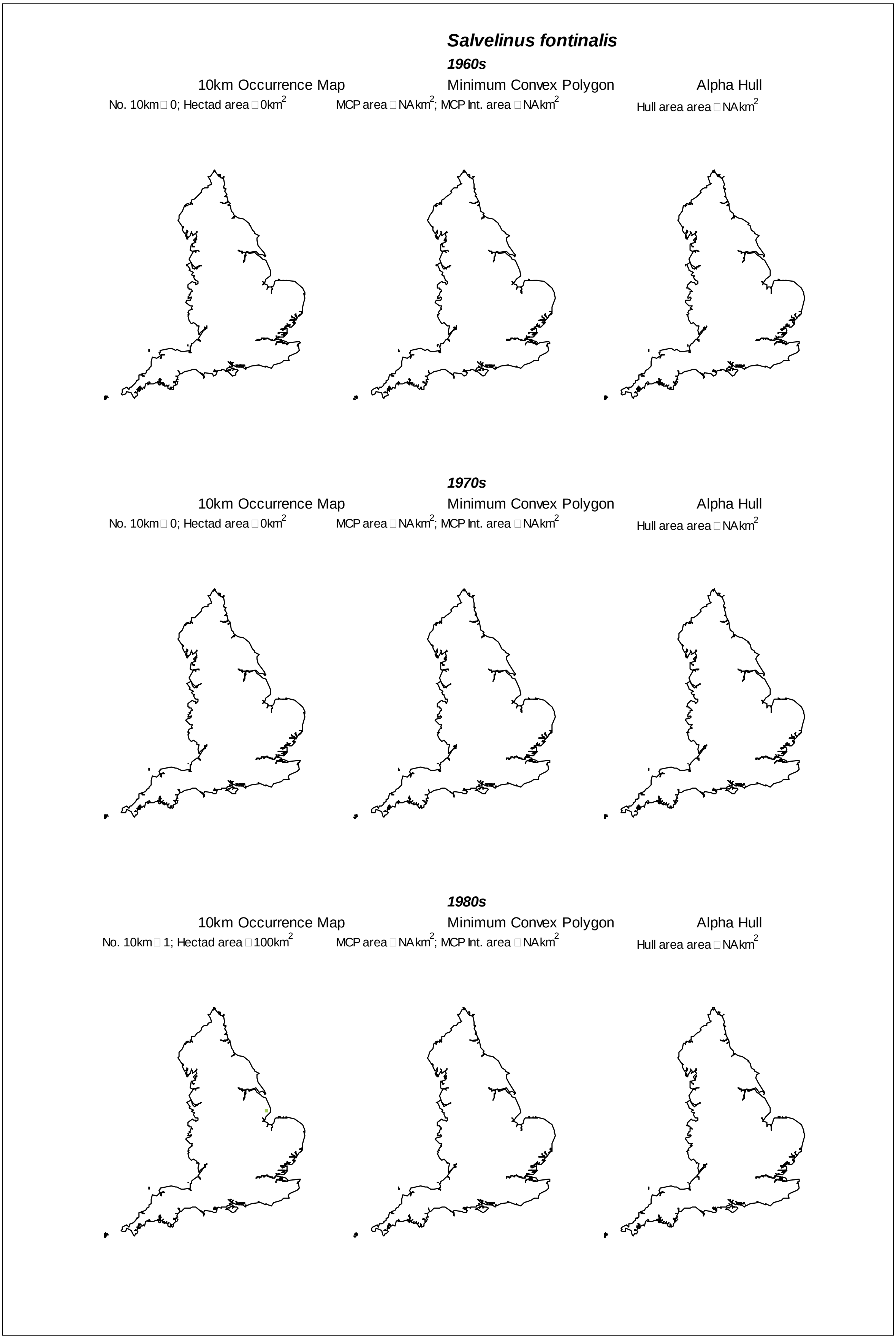


Figure C.72a Change in area of extent for *Salvelinus fontinalis* by decade (1960s to 1980s)

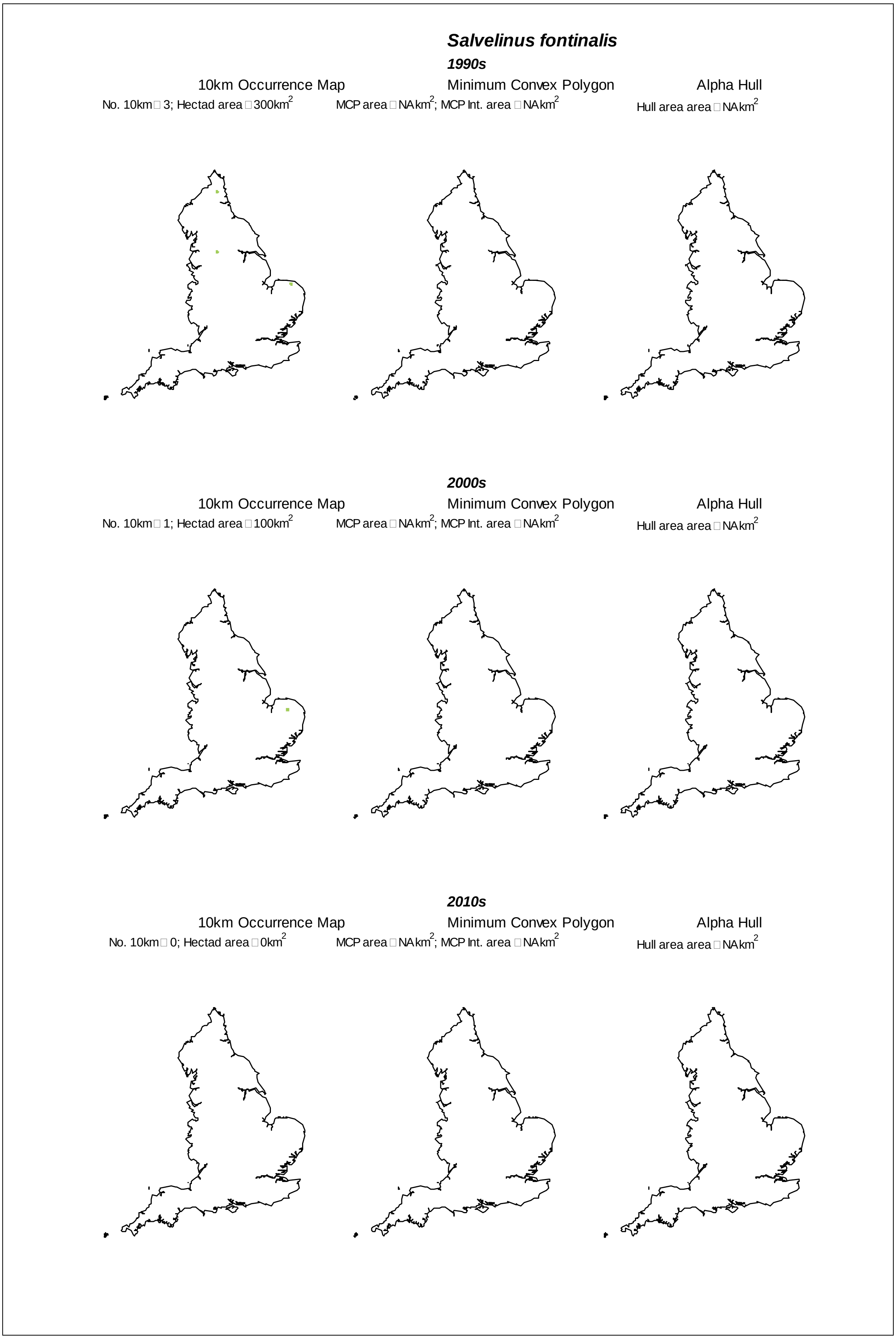


Figure C.72b Change in area of extent for *Salvelinus fontinalis* by decade (1990s to 2010s)

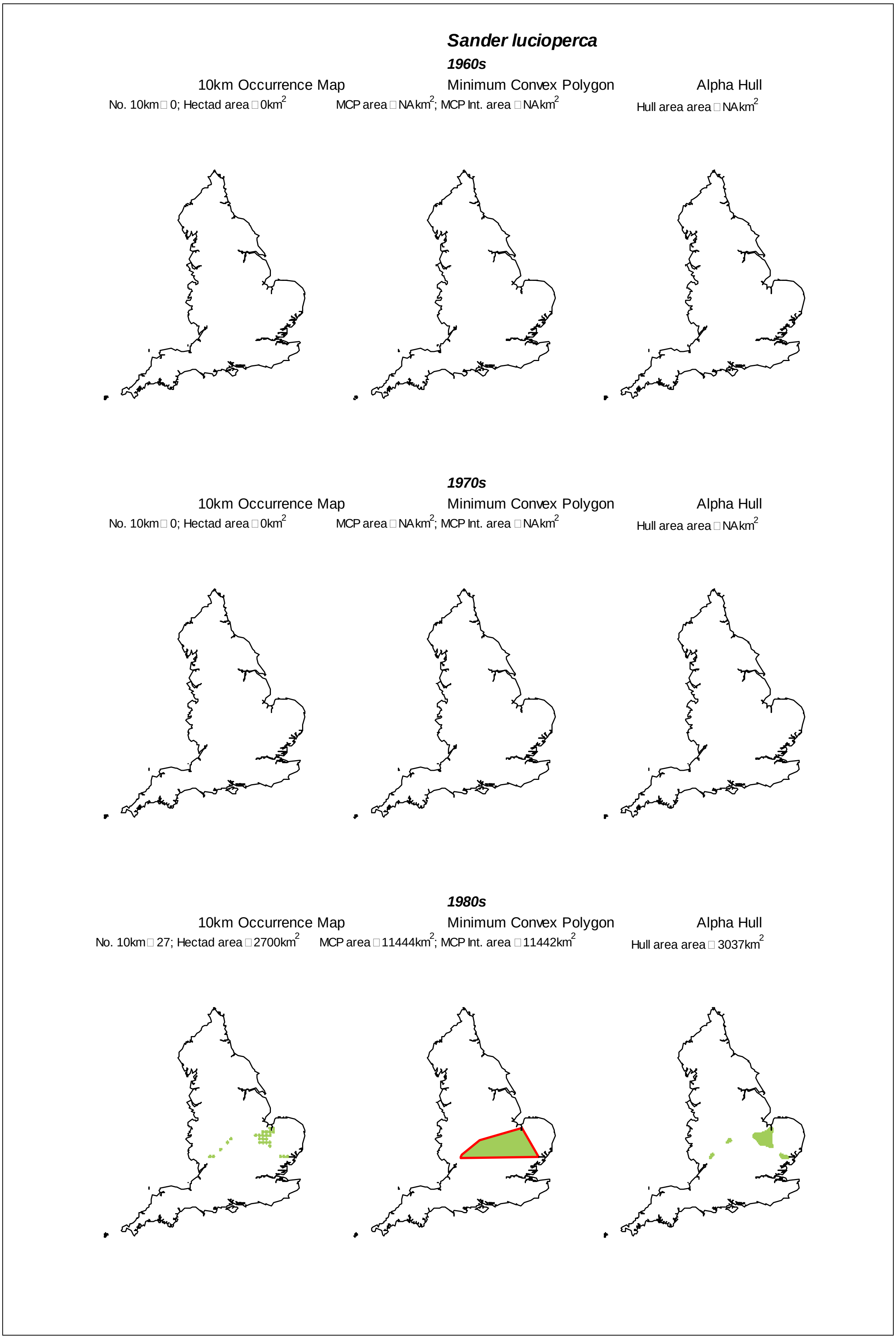


Figure C.73a Change in area of extent for *Sander lucioperca* by decade (1960s to 1980s)

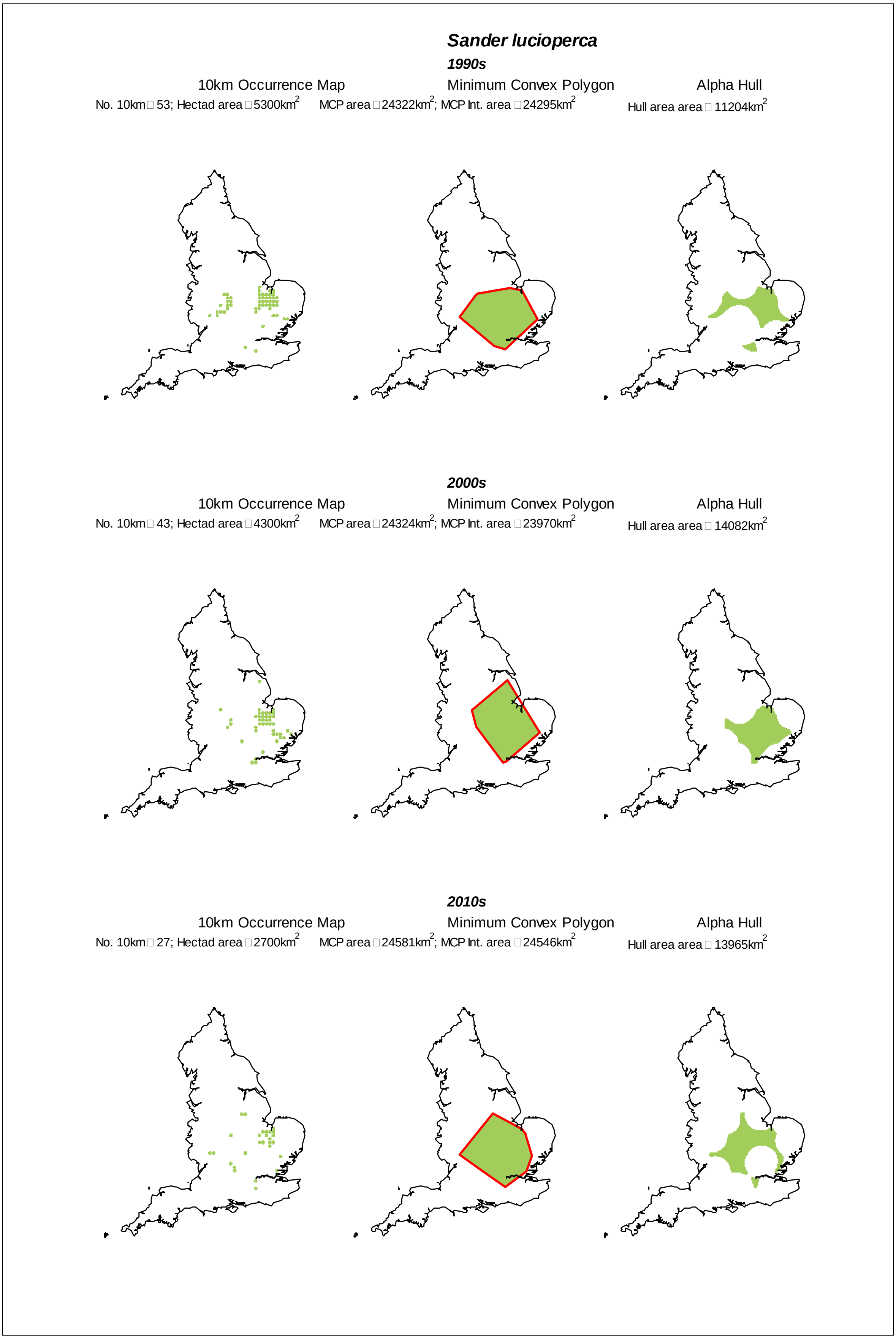


Figure C.73b Change in area of extent for *Sander lucioperca* by decade (1990s to 2010s)

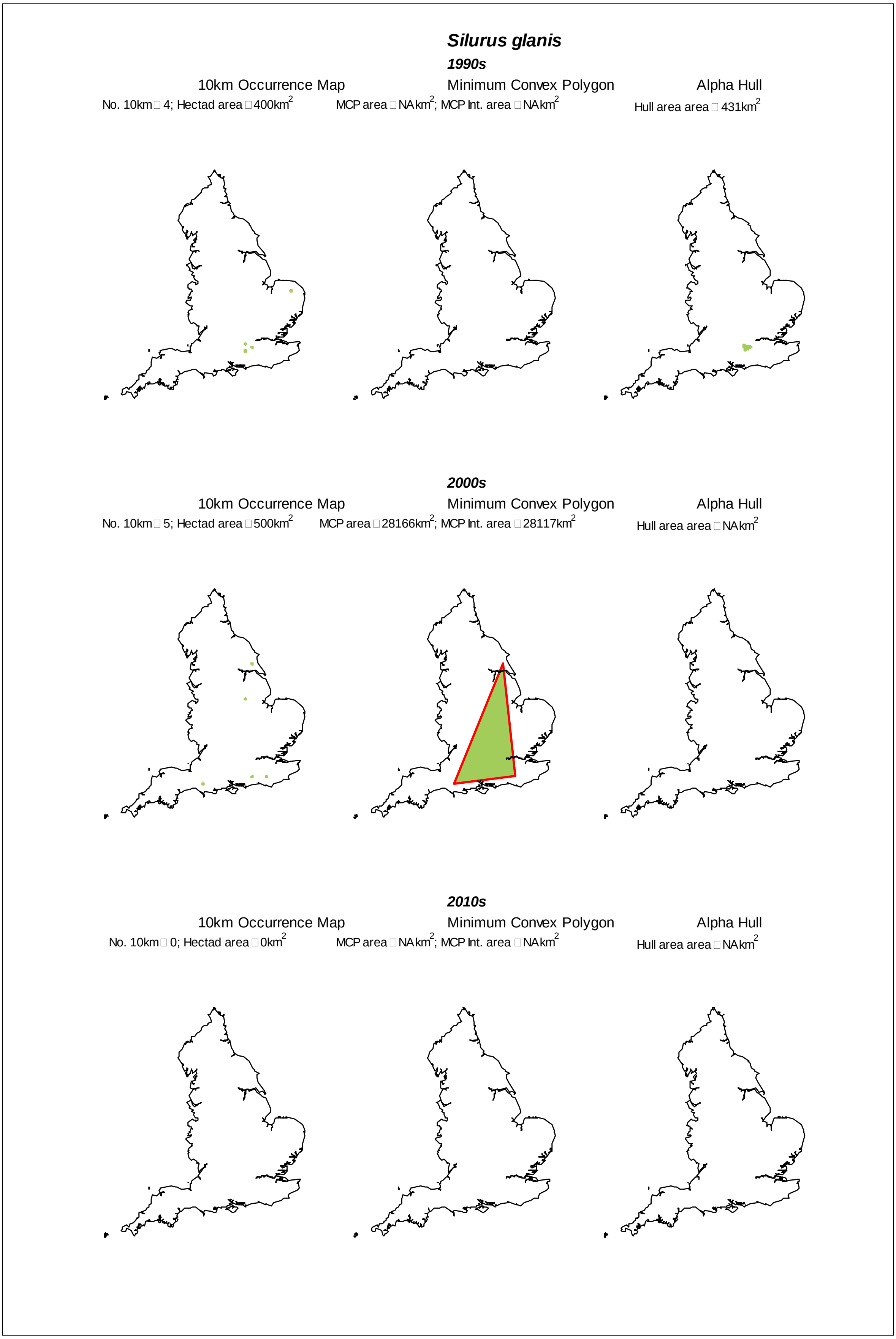


Figure C.74 Change in area of extent for *Silurus glanis* by decade (1990s to 2010s)

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