APPENDIX

CASE STUDY 1: *Celastrus orbiculatus*

Step 1: Data synthesis - Linking the invasion process to population demography

To compile all the data available on this species within its invasive range in North America, we first carried out a literature search, using the database *Web of Science*, of all the experimental and observational work published about *C. orbiculatus* in scientific journals (Table A-1). Information ranged from distribution data, e.g., presence and abundance, to population germination and establishment rates, and individual growth rates. We selected two response variables, presence and abundance of *C. orbiculatus*, that were based on geographically referenced data available through the Invasive Plant Atlas of New England (Mehrhoff et al. 2003; $N = 4200$ observations). This data set also provided the data source for constructing the model, i.e. information on the specific characteristics of each site (habitat type, canopy closure and soil moisture), and the fact that it is geographically-referenced allowed us to link each observation to climate and land use land cover (LULC) data (Ibáñez et al 2009*a,b*).

Next we organized this information by assigning each information source to a specific context, species, habitat or system (Foxcroft et al. 2011), and to each of the stages defined as part of the demographic framework (Table A1). We also kept track of the organismal or spatial scale at which information was collected.

Step 2: Data-model model integration using hierarchical models

We tried different models that varied in the level of information included, from a simple generalized linear model using only climate data to a more complex framework that included a large portion of the information found in the literature search. We used the Deviance Information

Criterion, DIC (Spiegelhalter et al. 2000) to select the model that best fit the data (Table A2). We describe below the final model used in the integrated assessment analysis.

Given our knowledge of the invasion, e.g., *C. orbiculatus* populations associated to particular landscape features (e.g., Albright et al. 2009, Kelly et al. 2009) and to specific habitats (Ibáñez et al. 2009*a,b*), we estimated the probability of the species being present at site *i*, π_i, as a function of the independent probabilities that determine propagule availability (*pr*), suitability of the site (*su*), germination (*ger*) and establishment (*est*). *Y* denotes the data, 0 if absent and 1 if present:

$$Y_i \sim Bernoulli(\pi_i)$$

$$\pi_i = pr_i su_i ger_i est_i$$

Propagule availability (*pr*) was calculated as a function of developed land around the sample point to reflect the association of *C. orbiculatus* populations with human development in the landscape. Beside the fix effect coefficients, α_*, we also included a spatial random effect (ω). This spatial effect had a double function here; it allowed us to account for any spatial autocorrelation in the data, and, as this species is still moving through the landscape and some of the absences in the data may not be true absences, this term also took into account that an area may not have been invaded yet even if the conditions were suitable for colonization (Ibáñez et al. 2009*a*):

$$logit(pr_i) = \alpha_1 + \alpha_2 \%\ developed\ land_i + \omega_i$$

Suitability of a site was modeled as a function of habitat type nested within climate, i.e., climate's impact on the suitability of a site depended on the type of habitat. Again this reflects information gathered from the literature search; open habitats reduced winter survival of first-year seedlings, while survival in later stages was greater under open canopies (Ellsworth et al.

2004*a*). The hierarchical structure allowed us to account for the fact that a change in temperature in closed canopy forests may have much less of an impact than similar changes in an open habitat:

$$logit(su_i) = \beta_0 + \beta_{1,k(i)} summer\ temperature_i + \beta_{2,k(i)} winter\ temperature_i +$$

$$\beta_{3,k(i)} annual\ precipitation_i + \beta_{4,k(i)} precipitation\ seasonality_i$$

$$+ \beta_{5,k(i)} summer\ precipitation_i$$

Here the fixed effects coefficients associated to each covariate, β, are estimated for each habitat type, k, so for example the effect of winter temperature can vary among habitat types. We then estimated these parameters from prior distributions with parameters, β_j and $\sigma^2_{\beta j}$, which represent the overall response of the species to that variable:

$$\beta_{j,k} \sim Normal\left(\beta_j, \sigma^2_{\beta j}\right)$$

This approach is useful because we can predict suitability of specific habitats using parameters $\beta_{j,k}$, or across all habitats, using β_j and $\sigma^2_{\beta j}$; at the same time, and we can compare the effects of climate variables among habitat types. When examining suitability, parameter $\sigma^2_{\beta j}$ accounts for the variability in the response found among habitat types.

Germination and establishment rates were not estimated, but values were drawn from distributions that reflect the outcome of previous experimental studies (Dreyer et al. 1987, Ellsworth et al. 2004*b*). Variability around those estimates was included in the analysis to help account for the different sources of uncertainty inherent in the presence/absence data:

$$ger_i \sim Uniform(0.55, 0.75)$$

$$est_i \sim Uniform(0.8, 1)$$

Percent cover– our proxy for proliferation – was recorded in six ordinal categories. To analyze these data, we then used a proportional odds model (McCullagh 1980, Cox 1995). Cover was analyzed as a function of both site suitability, as described above, and site conditions, i.e., light and soil moisture availability. Here again we used a hierarchical structure to estimate the effects of canopy closure nested within soil moisture level. Empirical work shows the influence of water and light availability on growth rates (Leicht and Silander 2006), and as these two variables do not act independently we decided to model the effects of light availability nested within the soil moisture level (*s*) of a particular site, i.e., the effects of a particular light level may depend on the availability of other resources, in this case water. The probability, μ, that a particular percent cover category, *g,* is recorded is estimated as:

$$logit(\mu_{i,g}) = (\lambda_0 + \lambda_{s(i)} canopy\ closure_i) su_i + \varepsilon_i$$

with fixed effect coefficients, λ, and random individual effects, ε.This way the effects of canopy closure, i.e., light, were estimated for each soil moisture category, *s* (Table A1).

All parameters specified in the model were estimated from distributions with non-informative parameter values, or priors:

-fixed effects coefficients

$$\alpha_*, \beta_*, \lambda_* \sim Normal(0,10000)$$

-random effects

$$\varepsilon_i \sim Normal(0, \sigma_\varepsilon^2)$$

$$\omega_i | \omega_{j,j\neq i} \sim Normal\left(\sum_j b_{ij}\omega_j, \sigma_\omega^2\right)$$

b_{ij} is an indicator function, = 1 if locations *i* and *j* are within a 10 km radius of each other, and = 0 otherwise (Ibáñez et al. 2009*a*).

-variances

$$1/_{\sigma_{\omega}^{2}} \sim Gamma(0.5,0.0005)$$

$$1/_{\sigma_{\varepsilon}^{2}} \sim Gamma(0.01,0.01)$$

$$\sigma_{\beta*} \sim Uniform(0,1000)$$

Models were run in OpenBUGS 1.4 (Thomas et al. 2006). Three chains were run to ensure convergence of the parameter estimates (100,000 iterations), and parameter estimate values were calculated after discarding the burn-in period (~25,000 iterations).

Step 3: Assessment of biological invasions

We evaluated the different models we tried (Table A2) by their performance predicting the data, but we did not validate them with independent data, as these are rare (and never available for potential invaded areas or future climates). We also analyzed the data by implementing a climate envelope colonization model that estimated the presence/absence data as a function of climate alone, and then comparing the climate envelope model estimates with our suitability estimates from the integrated assessment (Fig. A1). Parameter estimates are listed in Table A3.

We generated forecasts under different habitats and under two climate scenarios for the end of the century, from two climate models: CCSM3 and UKMO-HadCM3 (Fig. A2). Of the models participating in the IPCC AR4, these produced the two most disparate climate projections (most moderate and most extreme respectively (Wang 2005). We used the B1 scenario, which maintains current levels of carbon dioxide emissions.

TABLE A1. List of empirical and observational studies of *Celastrus orbiculatus* invasions in North America, and list of data sources used in the analysis. (*) Indicates information/data included in the final model.

Category	Information	Source
Species context-Demographic process		
Reproduction	Pollen viability 67%	Dreyer et al.1987
	Mean seed rain 168 seeds/m^2	Ellsworth et al. 2004*b*
	*Seed germination 59-82%	Dreyer et al.1987
	Seed viability 60%	Ellsworth et al. 2004*b*
	No seed bank	Ellsworth et al. 2004*b*
Dispersal	By birds, mammals, humans	Dreyer et al.1987
	Long distance by birds	Dreyer et al.1987
	Fruit traits that promote bird dispersal	Gosper et al. 2005
	Removal rate 76 % Dispersal rate 75%	Greenberg et al. 2001
	Root suckering	Dreyer et al.1987
Establishment	*Seedling emergence ~100%	Ellsworth et al. 2004*b*
	*Intact litter reduces emergence 20%	Ellsworth et al. 2004*b*
	*Thick litter reduces establishment, most abundant in evergreen stands	Facelli and Pickett 1991
Survival	Can establish at low light	Ellsworth et al. 2004*a*
	*First winter survival lower in open sites	Ellsworth et al. 2004*a*
	*Second year survival 25% lower at low light	Ellsworth et al. 2004*a*
Growth	Plastic growth responses	Clement et al. 1991
	*Greater growth in high light	Ellsworth et al. 2004*a*
	At low light higher growth rates than native *Celastrus*	Leicht-Young et al. 2007 Leicht and Silander 2006
	Similar population dynamics than the native community	Meiners 2007
	Longer growing season than native competitor *Vitis riparia*	Tibbetts & Ewers 2000
Habitat context-Susceptibility		

Habitat preferences	*Associated with low soil drainage and mesic environments	Kelly et al. 2009 McNab & Loftis 2002
	*Mainly found at forest edge habitats, scarce in forest interior	Ibáñez et al. 2009 Kuhman et al. 2010
	Less abundant in areas with disturbed forest floor, and wind-disturbed areas	McNab & Loftis 2002
	Associated with low *Quercus* abundance (drier, thick litter layer)	Kelly et al. 2009
Landscape configuration	*Associated with urban/developed areas	Ibáñez et al. 2009 Albright et al. 2009 Kuhman et al. 2010
	*Associated with mesic conditions	McNab & Loftis 2002 Pande 2007 Kuhman et al. 2010
Climate	*Associated with lower summer temperature	Ibáñez et al. 2009*a*
	*Associated with higher winter temperature	Ibáñez et al. 2009*a*
	*Associated with higher precipitation	Ibáñez et al. 2009*a*
	*Associated with higher summer precipitation	Ibáñez et al. 2009*a*
System context		
Legacies	*Past disturbances associated with current presences and higher cover	Kuhman et al. 2010
	Past land history associated with presence	Lundgren et al. 2004
Windows of opportunity	Synergistic expansion in the Southern Appalachians as hemlock wooly adelgid is creating newly suitable areas	Albright et al. 2009
	*Higher abundance and likelihood of presence in abandoned sites	Ibáñez et al. 2009*a,b*
	Higher presence in disturbed sites	Mosher et al. 2009
	Invasion triggered by logging disturbances	Silveri et al. 2001
Data sources		
Presence/absence and percent cover	*4200 geo-referrenced plots were surveyed in New England; plot data on habitat type, soil moisture, and canopy	IPANE-Mehrhoff et al. 2003

	closedness were also collected	
	*Percent cover levels (%): <1, 1-5, 6-25, 26-50, 51-75, and 76-100	Ibáñez et al. 2009*b*
	*Habitats considered: edge, deciduous and mixed forests, evergreen forests, open wetlands, closed canopy wetlands, marine influence, and abandoned	Ibáñez et al. 2009*a,b*
	*Soil moisture levels: xeric, mesic, saturated, and inundated	IPANE-Mehrhoff et al. 2003
	*Canopy closure levels (%): 0-25, 26-50, 51-75, and 76-100	Ibáñez et al. 2009*a,b*
Climate	*Long-term averages at a ~1km resolution associated with each surveyed plot	WorldClim-Hijmans et al. 2005
	*Variables considered: mean summer maximum temperature, mean winter minimum temperature, annual precipitation, summer precipitation, and precipitation seasonality	Ibáñez et al. 2009*a*
Land Use Land Cover (LULC)	*Percent of each land use category in a 10 km radius around each surveyed plot from 30x30 m resolution satellite data	2001 New England LULC data (NOAA 2007)
	*Categories used: developed, deciduous and mixed forests, evergreen forests, crops, scrub/shrubland, and pastures/grasslands	Ibáñez et al. 2009*a*

TABLE A2. List of selected models run to estimate *Celastrus* invasion potential, and associated Deviance Information Criterion (DIC). Models show an increase in complexity and amount of information included with decreasing DIC. The first model is representative of a niche model that includes only climate and presence/absence data. The final model selected was the one that better explained the presence/absence and cover data, and had the lowest DIC (bold). This model included most of the demographic data available and accounted for all phases of the invasion process.

Model	DIC
1. Climate only informing site suitability	3912
2. Climate informing site suitability, LULC data informing propagule availability, and germination and establishment rates included	3885
3. Climate informing site suitability through habitat type, LULC data informing propagule availability, and germination and establishment rates included	3819
4. Climate informing site suitability through habitat type, LULC data informing propagule availability and spatial random effects included, and germination and establishment rates included	3811
5. Climate informing site suitability through habitat type, LULC data informing propagule availability and spatial random effects included, and germination and establishment rates included. Percent cover data estimated as a function of site suitability, soil moisture and light	**3773**

TABLE A3. Parameter estimates from the best *Celastrus* model (posterior mean values (SD)). Bold values indicate fixed effect coefficients that are statistically significant (i.e., 95% CI, below, does not include zero).

Parameter	Mean(SD) 95%CI	Parameter	Mean(SD) 95%CI	Parameter	Mean(SD) 95%CI
α_0 intercept	-0.04(0.26) -0.53,0.5	$\beta_{3,5}$ closed-canopy wetland	-1.38(1.05) -3.16,0.41	σ_ε^2	733(64) 879.5,619.9
α_1 land developed	**2.22 (0.92)** **0.97,4.45**	$\beta_{3,6}$ marine	-0.67(1.74) -4.26,2.5	σ_ω^2	0.006(0.002) 0.012,0.004
β_0 intercept	**-0.42(0.07)** **-0.57,-0.28**	$\beta_{3,7}$ abandoned	**1.76(0.79)** **0.27,3.35**	$\sigma_{\beta 1}^2$ summer temp	0.33(0.16) 0.17,0.5
β_1 summer temp	-0.06(0.56) -1.19,1.08	β_4 prec seasonality	**-0.97(0.46)** **-1.9,-0.03**	$\sigma_{\beta 1}^2$ winter temp	0.33(0.11) 0.21,0.45
$\beta_{1,1}$ edge	**0.73(0.31)** **0.12,1.32**	$\beta_{4,1}$ edge	**-1.04(0.41)** **-1.86,-0.24**	$\sigma_{\beta 1}^2$ annual prec	0.67(0.41) 0.25,1.08
$\beta_{1,2}$ dec-mix forest	**-0.78(0.41)** **-1.6,-0.03**	$\beta_{4,2}$ dec-mix forest	**-1.46(0.5)** **-2.57,-0.6**	$\sigma_{\beta 1}^2$ prec seasonality	0.39(0.18) 0.2,0.58
$\beta_{1,3}$ ever forest	-0.85(0.58) -2.08,0.13	$\beta_{4,3}$ ever forest	-0.83(0.49) -1.72,0.15	$\sigma_{\beta 1}^2$ summer prec	0.3(0.21) 0.08,0.52
$\beta_{1,4}$ open wetland	**-1.24(0.6)** **-2.49,-0.21**	$\beta_{4,4}$ open wetland	**-1.5(0.55)** **-2.71,-0.58**		
$\beta_{1,5}$ closed-canopy wetland	0.20(0.66) -0.99,1.54	$\beta_{4,5}$ closed-canopy wetland	-0.91(0.71) -2.32,0.55		
$\beta_{1,6}$ marine	1.45(0.87) -0.04,3.27	$\beta_{4,6}$ marine	-1.37(0.79) -1.49,1.53		
$\beta_{1,7}$ abandoned	0.27(0.56) -0.87,1.38	$\beta_{4,7}$ abandoned	-0.13(0.79) -1.49,1.53		
β_2 winter temp	**2.35(0.44)** **1.53,3.29**	β_5 summer prec	0.39(0.47) -0.48,1.45		
$\beta_{2,1}$ edge	**1.8(0.33)** **1.17,2.44**	$\beta_{5,1}$ edge	-0.27(0.34) -0.97,0.35		
$\beta_{2,2}$ dec-mix forest	**2.85(0.53)** **1.93,3.91**	$\beta_{5,2}$ dec-mix forest	0.05(0.37) -0.97,0.35		
$\beta_{2,3}$ ever forest	**3.1(0.65)** **2,4.43**	$\beta_{5,3}$ ever forest	0.18(0.38) -0.52,0.91		
$\beta_{2,4}$ open wetland	**2.2(0.59)** **1.03,3.41**	$\beta_{5,4}$ open wetland	**0.96(0.48)** **0.09,1.89**		
$\beta_{2,5}$ closed-canopy wetland	**2.56(0.71)** **1.35,3.98**	$\beta_{5,5}$ closed-canopy wetland	1.37(0.85) -0.02,3.16		
$\beta_{2,5}$ marine	**2.1(0.67)** **0.87,3.36**	$\beta_{5,6}$ marine	1.09(0.91) -0.39,3.18		
$\beta_{2,7}$ abandoned	**1.63(0.52)** **0.64,2.64**	$\beta_{5,7}$ abandoned	-0.55(0.56) -1.75,0.41		
β_3 annual prec	-0.46(0.81) -2.17,1.02	λ_0 intercept	**43.5(3.39)** **38.5,49.3**		
$\beta_{3,1}$ edge	**0.83(0.43)** **0.05,1.67**	λ_1 xeric	**-0.26(0.19)** **-0.68,0.1**		
$\beta_{3,2}$ dec-mix forest	-0.96(0.56) -2.01,0.11	λ_2 mesic	**-0.33(0.07)** **-0.46,-0.2**		

$\beta_{3,3}$ ever forest	-1.19(0.65) -2.42,0.02	λ_3 saturated	**-0.77(0.2)** **-1.19,-0.37**
$\beta_{3,4}$ open wetland	**-1.29(0.64)** **-2.59,-0.01**	λ_4 inundated	**-4.89(2.96)** **-10.4,-0.2**

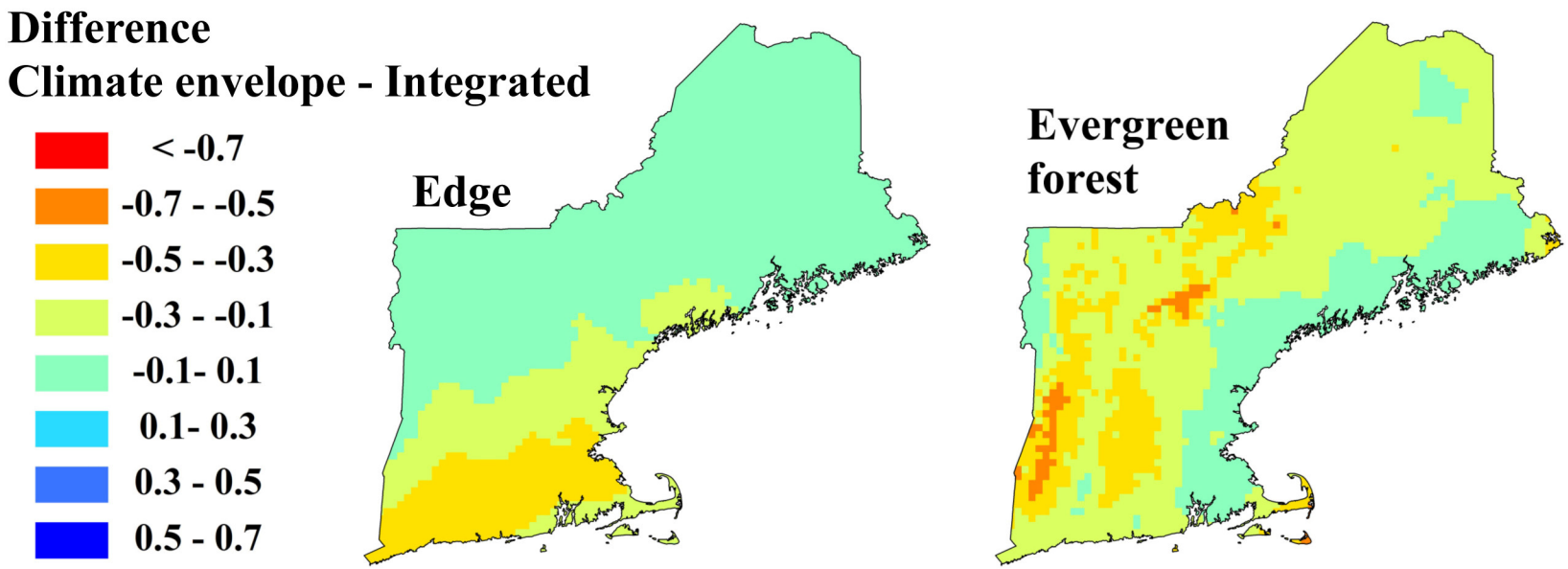

FIG. A1. Differences between climate envelope (DIC 3912) and integrated (DIC 3773) model predictions of the likelihood of colonization for edge and evergreen forest habitats. For edge habitats, which are more suitable for *Celastrus*, the integrated model predicts greater colonization potential than the null model, but only in the southern areas of the region, for evergreen forest, the integrated model predicts greater colonization potential than the null model for most of the region, in both cases better fitting the data.

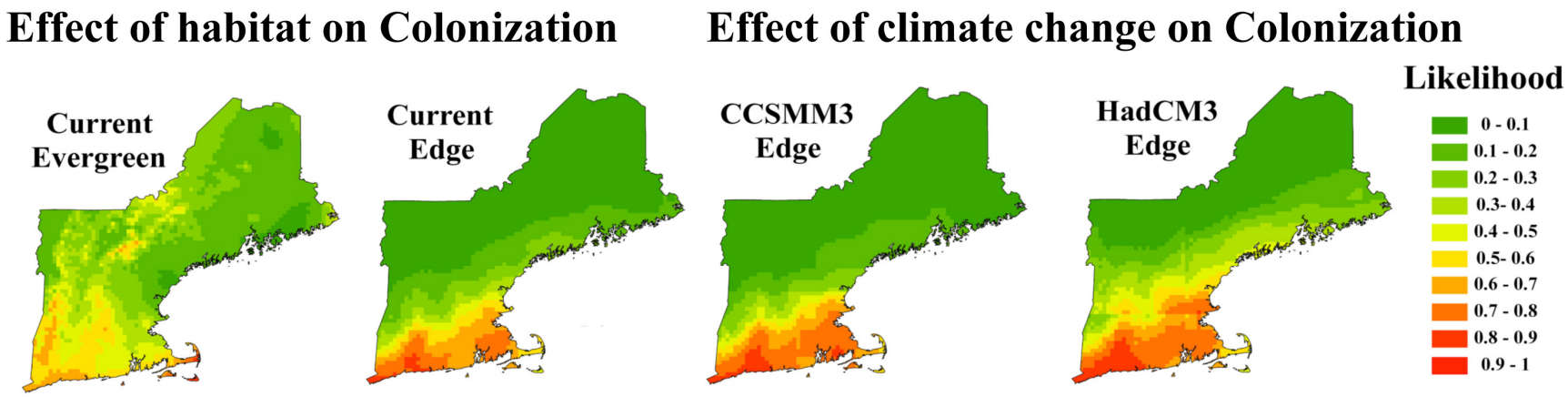

FIG. A2. Left panels: likelihood of colonization under current climate and landscape configuration at two different habitats showing how the combination of climate and habitat is not additive, and strongly modulates the suitability of a site. Right panels: likelihood of colonization under two future climate scenarios, only under the most extreme climate predictions, HadCM3 model, we predict substantial changes with respect to current conditions.

CASE STUDY 2: *Mytilus galloprovincialis*

Steps 1: Data synthesis - Linking the invasion process to population demography

We gathered information on species and habitat context from 62 observational and experimental studies of *M. galloprovincialis*, which provided data on reproduction, dispersal, recruitment, juvenile and adult survival, and larval, juvenile, and adult growth rates (Table A4). The two response variables we included in the analysis were adult survival and adult growth rates, as proxies for potential to colonize and proliferate, respectively. We used the geo-referenced points associated with each of the studies to pair data on sea surface temperature and chlorophyll *a* content (Feldman and McClain 2012) with demographic records.

Step 2: Data-model model integration using hierarchical models

To estimate the suitability of a site as a function of environmental variables and growth as a function of resources, we used tidal zones, maximum sea surface temperatures, and maximum chlorophyll levels as the explanatory variables in our model. In this case, given the small number of predictive variables and of observations available (~20 for each sub-model), we considered a smaller number of models (Table A5). Other models run with variations of these predictors, e.g., average or minimum temperature, did not generate substantially different results.

Observational studies have recorded mean dispersal distances of ~30 km (Becker et al. 2007, Carson et al. 2010), and maximum dispersal of ~60–100 km (Gilg and Hilbish 2003, Branch and Steffani 2004). We used that information to fit an exponential decay function ($\rho e^{-\rho x}$) that estimated the probability of colonization as a function of dispersal from larval sources, with a mean (1/ρ) of ~30 km. Thus, *dispersal* probability was estimated a priori (before running the model) and varied from 1, next to the larval source, to almost 0 at 100 km, which would still

allow for rare long-distance dispersal events and reflect the patterns of dispersal that have been reported.

We then used the survival and growth data to estimate the relative performance of this species under varying environmental conditions and across habitats. As data available came from studies that were run for different periods of time, we standardized the data by calculating daily survival rates, '*Survival'* (N = 24 observations). Then for each observation i:

$Survival_i \sim Normal(Sur_i, \sigma_i^2)$ limited between 0–1 (we also tried using a *Beta* distribution but a *Normal* with limits 0–1 worked better)

$$logit(Sur_i) = \gamma_{z(i)} + \gamma_5 * temperature_max_i$$

A different slope, $\gamma_{1\text{-}4}$, was estimated for each of the four tidal zones z (sub-tidal and low-, mid- and high-shore), to reflect differences in suitability among the zones because of factors not accounted for (e.g., predation). Parameters γ were estimated from non-informative prior distributions, $\gamma \sim Normal(0,10000)$. We used reported values (standard errors and standard deviations around survival) to estimate the variance associated to each observation, σ_i^2.

Growth rate (*Growth*; mm/day), our proxy for *proliferation* (as growth is related to competitive ability), was also modeled as a function of environmental covariates (N = 20 observations):

$Growth_i \sim Normal(Gr_i, \sigma_i^2)$ limited to > 0 (we also tried using a *logNormal* distribution, but a *Normal* captured the different growth trends across habitats better)

$$Gr_i = \alpha_1 + \alpha_2 * Chlorophyll_max_i + \beta_{z(i)} * temperature_max_i$$

Parameters were then estimated from non-informative prior distributions, $\alpha_{*} \sim Normal(0,10000)$. To take into account the effect of temperature on growth rates at each of the tidal zones (z), we followed a hierarchical approach:

$$\beta_z \sim Normal(\beta m, \sigma_\beta^2)$$

$$\beta m \sim Normal(0,10000) \text{ and } \sigma_\beta \sim Uniform(0,1000)$$

And as we did in the survival submodel, we used reported values (standard errors and standard deviations around growth rates measurements) to estimate the variance associated with each observation, σ_i^2.

The model was run in OpenBUGS 1.4 (Thomas et al. 2006). Three chains were run to ensure convergence of the parameter estimates (50,000 iterations), and parameter estimates values were calculated after discarding the burn-in period (~10,000 iterations).

Step 3: Assessment of biological invasions

In this case study, we were again not able to validate our final model outcome with independent data, although the best model was selected based on the models' performance predicting the data (Table A5). From the parameters' estimates (means, variances and covariances; Table A6), we assessed the potential for invasion under several maximum chlorophyll *a* levels (low: 2, mid: 50, high: 100 mg m^{-3}) and as a function of distance to source (0 to 50 km) and of maximum water temperature (15 to 25 °C, see Fig. 5 in main text).

To represent the outcomes we standardized our estimates of daily rates of growth, *proliferation*, to range between 0 and 1, with zero representing lowest rates recorded and thus lowest potential for proliferation. We also estimated the complex pattern of uncertainty associated with our

predictions (Fig. A3), which in general illustrated that uncertainty was much lower for the sub-tidal zone than for the intertidal habitats, making predictions for the sub-tidal habitat most reliable.

We also compared our estimates of colonization with those from a climate envelope colonization model where only survival data were analyzed as a function of water temperatures and without differentiating among the four intertidal habitats, $logit(Sur_i)=\gamma_1 + \gamma_2 * temperature_max_i$ (Fig. A4).

TABLE A4. Empirical and observational studies of *Mytilus galloprovincialis* invasions from around the world. (*) Indicates information/data included in the final model.

Category	Information	Source
Species context-Demographic process		
Reproduction	Spawning season	
	– spring/summer	Bownes and McQuaid 2009
	- Spring/Summer, second spawn in Nov.	Doherty et al. 2009
	- Spring/Summer	Gilg et al. 2007
	- Late Summer	Petes et al. 2007
	Larval survival rates (caged larvae in field) 0.34± 0.47% survival	Becker et al. 2007
Recruitment rate	South Africa: sheltered shore ranged from 2.7 to 8.25 individuals m^{-2} yr^{-1}	Branch et al. 2008
	Semi-exposed shore ranged from 10.9 to 111.81 individuals m^{-2} yr^{-1}	Branch et al 2008
	Exposed shore ranged from 52.7 to 206.7 individuals m^{-2} yr^{-1}	Branch et al 2008
Larval Growth Rate	Larval growth rate ranges from 5 to 8.6µm day^{-1} depending on food availability	Phillips 2002
	Size at settlement ranges from 240-282µm depending on food	Phillips 2002
Survival rate (post-settlement juvenile)	South Africa: survival through 6 days on low shore 87.8± 15.57%, survival through 6 days on mid shore was 52.35± 29.04%, high shore was 83.02± 33.1%	Bownes and McQuaid 2009
Growth rate (post settlement)	North America: growth rate ranged from 2 to 35 µm day^{-1}	Phillips 2002
	South Africa: 3.5 to 143µm day^{-1}	Bownes and McQuaid 2009
Survival rate (adult)	*South Africa: low shore survival ranges from 20.6 to 56.8% up to 112 days, mid shore survival ranges from 41.8 to 66% up to 112 days, high shore survival ranges from 41.01 to 59.2%	Bownes and McQuaid 2010

	*New Zealand: daily mortality rate (over 134 days) estimated as -0.4± 0.097% day^{-1} and -0.38± 0.074% day^{-1} (1SE)	Menge et al. 2007
	*New Zealand: surviving proportion after 117-128 days: 42.9± 13.1% and 63.6± 4% (1SE).	Menge et al. 2007
	*South Africa: average monthly survival rate ranged from 85.7 to 98.34% at bay site, ranged from 84.7 to 95.9% at wave-exposed site	Nicastro et al 2010*a*
	*South Africa: cumulative survival after 1 year ranged from 68.6± 10% to 73± 6.6% in high zone, cumulative survival in mid zone ranged from 63.8± 14.6% to 67.3± 14.6% in mid zone	Rius and McQuaid 2009
	*Lab study: cumulative survival after 9 weeks in water or air/water ranged from 50.2% to 98.2%	Schneider 2008
	*Canada: survival after 3 months: 75% and 100% in subtidal	
	*USA: survival after 3 months: 90.4± 6.6% to 97.2± 0.64%	Shields et al. 2008
	*South Africa: monthly survival rate tracked over 6 months, ranged from 51.1± 36.6% to 95.4± 5.2%	Shinen and Morgan 2009
	*South Africa: survival rate after 7 months was 38.08% high shore, 5.9% mid shore, 0.7% low shore	Zardi et al. 2008
		Zardi et al. 2006
Growth rate (adult)	*South Africa: 0.0467± 0.036 mm day^{-1} in summer	Bownes and McQuaid 2010
	*South Africa: 0.057± 0.033 mm day^{-1}	Bownes and McQuaid 2010
	*France: 0.0538 mm day^{-1}	Kopp et al. 2005
	*New Zealand: 0.001± 0.0017 mm day^{-1} at Boulder Bay, 0.0424± 0.0465 mm day^{-1} at Box Thumb	Menge et al. 2007
	*New Zealand: high zone range 0 to 0.010 mm day^{-1}, low zone range 0.014 to 0.050 mm day^{-1}	Petes et al. 2007
	*USA: 0.008 to 0.044 mm day^{-1} depending on water + air	Schneider 2008

	temperatures (more data available)	
	*USA: field growth rates range from 0.0125 to 0.019 mm day^{-1}	Shinen and Morgan 2009
	*South Africa: growth rate at upwelling sites ranged from 0.0047 to 0.044 mm day^{-1}	Xavier et al 2007
	*South Africa: growth rate at non-upwelling sites ranged from 0.0005 to 0.043 mm day^{-1}	Xavier et al 2007
	*Italy: Scope for growth (J hr^{-1} available for growth + repro) at 14.67± 10.7 at 5m depth, 12.9± 12.43 J hr^{-1} at 15m depth	Sara and Pusceddu 2008
Habitat context-Susceptibility		
Water Temperature	*Lab: mussels acclimated at 14°C experienced heart failure at 26.5, 28.3, 25.7°C water temp for 22,28,34ppt acclimation salinities, respectively. Mussels acclimated at 21°C experienced heart failure at 29.8, 30.6, 30.8°C water temp for 22,28,34ppt acclimation salinities, respectively.	Braby and Somero 2006*a*
Salinity	Lab: acclimation at 22ppt produced lower critical salinity value of 8.85ppt, higher acclimation salinities had higher critical values.	Braby and Somero 2006*a*
Desiccation	*South Africa – survival after 4 days in air at 17°C + 60%RH was 96.8± 3.8% (1SD)	Nicastro et al. 2010
Anoxia	Lab: mussels begin dying after ~8 days in anoxic water at 20°. Switch to anaerobic metabolism occurs below oxygen concentrations of ~3ppm	De Zwaan et al 1991
Wave exposure	*Survival is generally poor at wave-exposed sites	Nicastro et al. 2010*b*, Rius and McQuaid 2006, Zardi et al. 2008
	*On wave-exposed shores, survival is greater at higher shore heights	Zardi et al. 2006
Temperature	*Survival is better than native *M.*	Schneider and Helmuth 2007,

	trossulus or *Perna* at sun-exposed (hot) sites.	Petes et al 2007
	*Overall survival is lowest in sun-exposed high shore sites (in protected bays)	Schneider and Helmuth 2007
	*Growth and survival are highest in subtidal conditions (absent predators)	Schneider 2008
System context	-	
Dispersal	*Disperse 20-30km before settlement	Becker et al. 2007
	*Mean dispersal distance 37(± 27.6) km, 28-42% self-recruitment within a site	Carson et al. 2010
	*90% of larvae settle within 12-20km of source. Average annual increase in range 42 km yr^{-1} eastward, 19km yr^{-1} westward	McQuaid and Phillips 2000
	*Average northward spread of 115 km yr^{-1}	Branch and Steffani 2004
	*Mean dispersal distances 24.3 km eastward (35 km max), 30 km northward (64 km max)	Gilg and Hilbish 2003
	-	

TABLE A5. Selected models run to estimate *Mytilus* potential for invasion. The final model best explained the survival and growth data, with the lowest DIC (bold).

Model	DIC
1. *Growth:* $\alpha + \beta_{z(i)} * temperature_max_i$	320
2. *Growth:* $\alpha_1 + \alpha_2 * Chlorophyll_average_i + \beta_{z(i)} * temperature_max_i$	285
3. *Growth:* $\alpha_1 + \alpha_2 * Chlorophyll_max_i + \beta_{z(i)} * temperature_max_i$	**251**

TABLE A6. Parameter estimates for the best *Mytilus* model (mean posterior values (SD) and 95% credible intervals (below)). Bold values indicate statistically significant fixed effects.

Parameter	Mean(SD) 95%CI	Parameter	Mean(SD) 95%CI
γ_1 intercept subtidal	9.04(0.04) 8.9,9.13	α_1 intercept	-0.01(0.01) -0.03,0.008
γ_2 intercept low-shore	3.11 (0.04) 3,3.2	α_2 Chlorophyll max	**0.0003(0.00008)** **0.0002,0.0005**
γ_3 intercept mid-shore	2.919(0.007) 2.91,2.93	β_m temperature max	0.001(0.04) -0.0009,0.003
γ_4 intercept high-shore	3.91(0.03) 3.84,3.98	β_1 subtidal	**0.002(0.0006)** **0.001,0.003**
γ_5 temperature max	**0.103(0.01)** **0.102,0.0104**	β_2 low-shore	**0.0015(0.0006)** **0.0012,0.003**
		β_3 mid-shore	0.0007(0.0006) -0.0004,0.002
		β_4 high-shore	0.0008(0.0006) -0.0004,0.002
		σ^2 *beta-temperature*	0.000001(0.000001) 0.0000002,0.00003

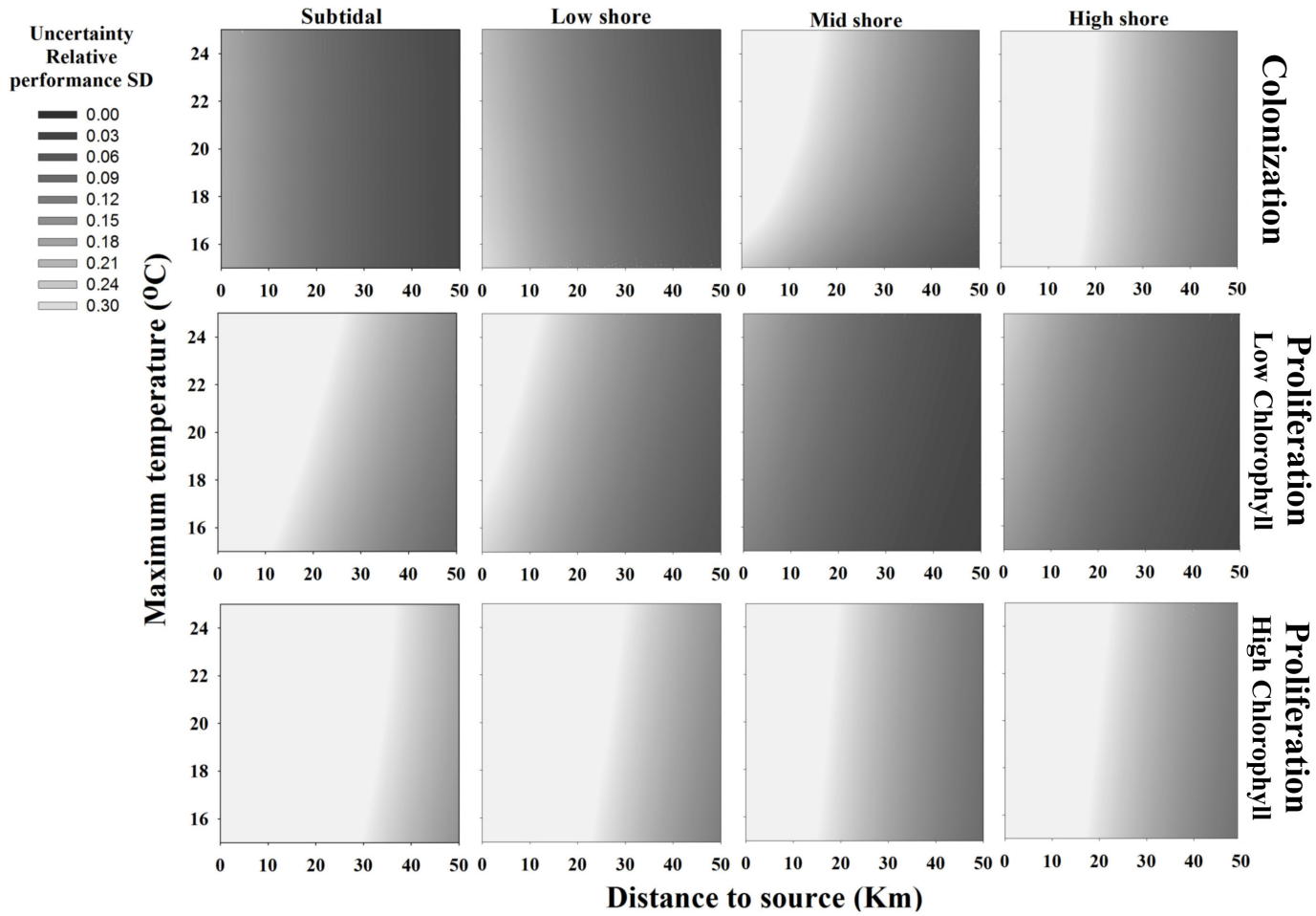

FIG. A3. Uncertainty estimates, shown as standard deviations around the mean posterior values for colonization (top panels) and proliferation (middle and bottom panels) associated with *M. galloprovincialis* invasions. Estimates are represented as a function of distance to larval sources and maximum sea surface temperature for each tidal zone, and of varying maximum chlorophyll *a* level (for spread estimates only).

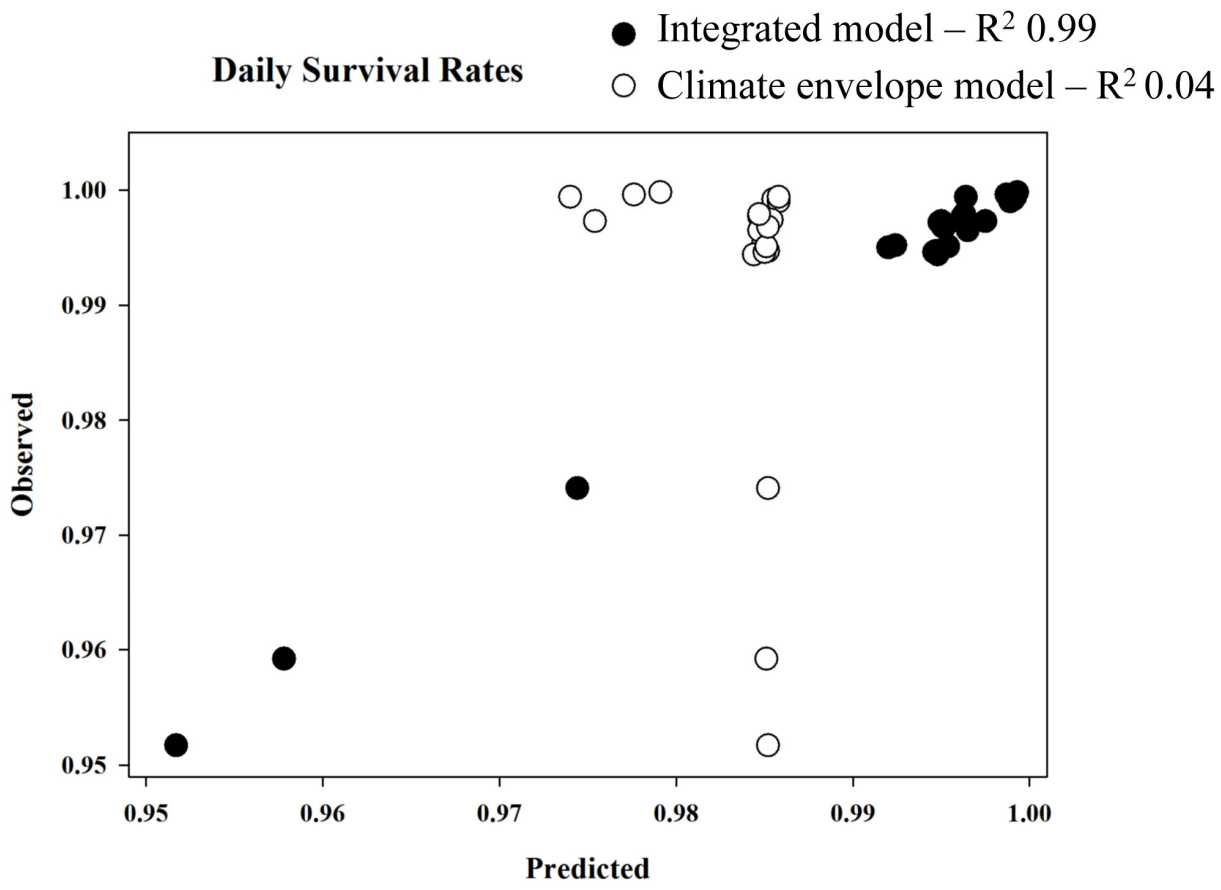

FIG. A4. Graphical representation of predicted *vs.* observed survival rates values for the climate envelope colonization model (open circles) and the integrated model (black circles).

CASE STUDY 3: *Orconectes rusticus*

Steps 1: Data synthesis - Linking the invasion process to population demography

To illustrate how we can work with varying levels of information, for this case study we used data from only one published study (Olden et al. 2011). This particular study does not include demographic data; the paper only provides presence/absence data for the invader and a measure of lake's vulnerability to the invasion based on presence/absence data for two native crayfish species before and after the introduction. This publication also reports the number of boat landings per lake, the type of water body (seepage or drainage), and a proxy for resource availability (Secchi disk depth – a measurement of water clarity). We identified winter minimum and summer maximum average air temperatures for each lake (PRISM Climate Group 2012). Temperature and resource availability data were standardized (ranging from -1 to 1) to facilitate comparisons of the parameter estimates. We tried several models and combinations of explanatory variables and report the best model fitting the data (Table A7).

Step 2: Data-model model integration using hierarchical models

Following Olden et al. (2011) we estimated probability of *dispersal* as a function of human visitation (boat landings) and included a spatial random effect. For each lake *i*:

$$logit(Dispersal_i) = \alpha_1 + \alpha_2 * landings_i + \omega_i$$

the fixed effect coefficients were estimated from non-informative prior distributions $\alpha_* \sim Normal(0,10000)$. Here again, as we were using presence/absence data of a species that is still moving into an area, we added a spatial random effect, ω_i, to reflect spatial autocorrelation

in the data and the fact that this species may not be in equilibrium with the environment (i.e., some absences may represent false absences):

$$\omega_i|\omega_{j,j\neq i}\sim Normal\left(-\left(\phi d_{ij}\right)^{\kappa},\sigma^2\right)$$

where d_{ij} is the distance between two locations, ϕ is the rate of decay estimated from $\phi\sim Gamma(0.1,0.1)$, the smoothing parameter, κ, was set up to 1, and $1/\sigma^2\sim Gamma(0.01,0.01)$.

Suitability of the site, our proxy for *colonization,* was calculated as:

$$logit(Suitability_i)=\beta_{wb(i)}+\beta_3*winter\ temperature_min_i+\beta_4*summer\ temperature_max_i$$

and fixed effect coefficients were estimated as $\beta_*\sim Normal(0,10000)$, with a different intercept for each water body type, β_{wb}. We then estimated the likelihood of *O. rusticus* being present as:

$$Presence_i\sim Bernoulli(Dispersal_i*Suitability_i)$$

We finished the analysis by using the reported vulnerability assessment to estimate the potential for *proliferation* of *O. rusticus*. Vulnerability rankings (0–1) for each lake were used as an estimate of the competitive ability of this species to displace congeneric species, and were estimated as combination of the resources available (Secchi disk depth) and of the suitability of the site (estimated in the previous step).

$Vulnerability_i\sim Normal(Vul_i,\sigma^2)$ limited between 0–1 (we also tried using a *Beta* distribution but convergence of the parameters in a *Normal* with limits 0–1 was better)

$$logit(Vul_i)=(\gamma_1+\gamma_2*Secchi_disk_depth_i)*Suitability_i$$

Parameters were then estimated from non-informative prior distributions, $1/\sigma^2\sim Gamma(0.01,0.01)$, $\gamma_*\sim Normal(0,10000)$.

The model was run in OpenBUGS 1.4 (Thomas et al. 2006). Three chains were run to ensure convergence of the parameter estimates (50,000 iterations), and parameter estimate values were calculated after discarding the burn-in period (~25,000 iterations).

Step 3: Assessment of biological invasions

Using the parameter estimates (means, variances and covariances; Table A8), we assessed the potential for invasion under several scenarios that account for an increase in human visitation, temperature and resources (Fig. 7 main text). We also report the uncertainty associated with each estimate (Fig. A5). The *dispersal* and *proliferation* phases had the greatest uncertainty. This high level of uncertainty suggests that, even if the estimated likelihood of invasion is low in a given location, the possibility of invasion should be accounted for when considering future management plans.

We also compared the integrated model's estimates of colonization with those of a climate envelope model that only estimated colonization as a function of the climatic variables:

$$Presence_i \sim Bernoulli(Suitability_i)$$

and

$$logit(Suitability_i) = \beta_1 + \beta_2 * winter\ temperature_min_i + \beta_4 * summer\ temperature_max_i.$$

The climate envelope model had little predictive ability, with all means around 0.5 and standard deviations close to 0.5 (Fig. A6).

TABLE A7. Some of the models run to estimate invasion potential of *O. rusticus*. Models differ in the way that suitability and the availability of resources were combined to model lake vulnerability. The final model selected was the one that best explained the presence/absence data, with the lowest DIC (bold).

Model	DIC
1. Vulnerability: $\gamma_1+\gamma_2$**Secchi_disk_depth* No spatial random effects	401
2. Vulnerability: $\gamma_1+\gamma_2$**Secchi_disk_depth*+γ_2**Suitability* No spatial random effects	375
3. Vulnerability: ($\gamma_1+\gamma_2$**Secchi_disk_depth)*Suitability* No spatial random effects	337
4. Vulnerability: ($\gamma_1+\gamma_2$**Secchi_disk_depth)*Suitability* Included spatial random effects in *colonization*	**324**

TABLE A8. Parameter estimates from the best *Orconectes* model (mean posterior values (SD) and 95% credible intervals). Bold values indicate fixed effect coefficients that are statistically significant (i.e., 95% CI does not include zero).

Parameter	Mean(SD) 95%CI	Parameter	Mean(SD) 95%CI
α_1 intercept	-1.58 (0.28) -2.11,-1.02	β_3 winter temperature	0.08 (0.1) -0.07, 0.27
α_2 boat landings	**0.15 (0.08)** **0.017,0.34**	β_4 summer temperature	**-1.95(0.35)** **-2.67,-1.34**
ϕ rate of decline	0.00005(0.00003) 0.000008,0.0001	γ_1 intercept	-1.5(0.07) -1.798,-1.4
σ^2 *spatial*	0.68(0.19) 0.29,1.5	γ_2 Secchi disk depth	-0.11 (0.05) -0.22,0.004
β_1 drainage lake	3.87(1.01) 11.7,151.3	σ^2 *vulnerability*	0.01(0.009) 0.008,0.012
β_2 seepage lake	5.69(1.48) 2.81,8.34		

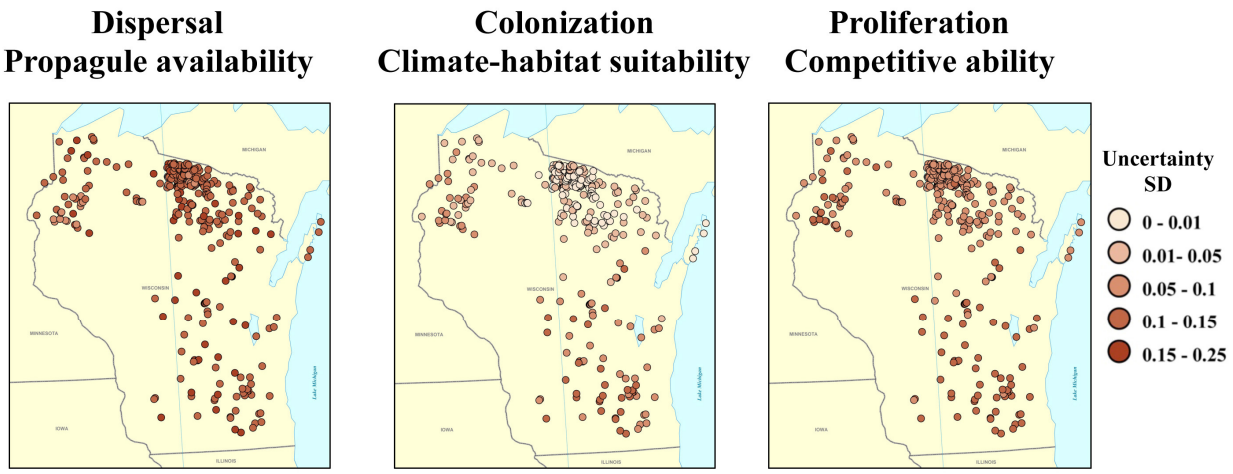

FIG. A5. Uncertainty estimates for each of the phases of *Orconectes* invasion, as standard deviations around the mean posterior values from the integrated model.

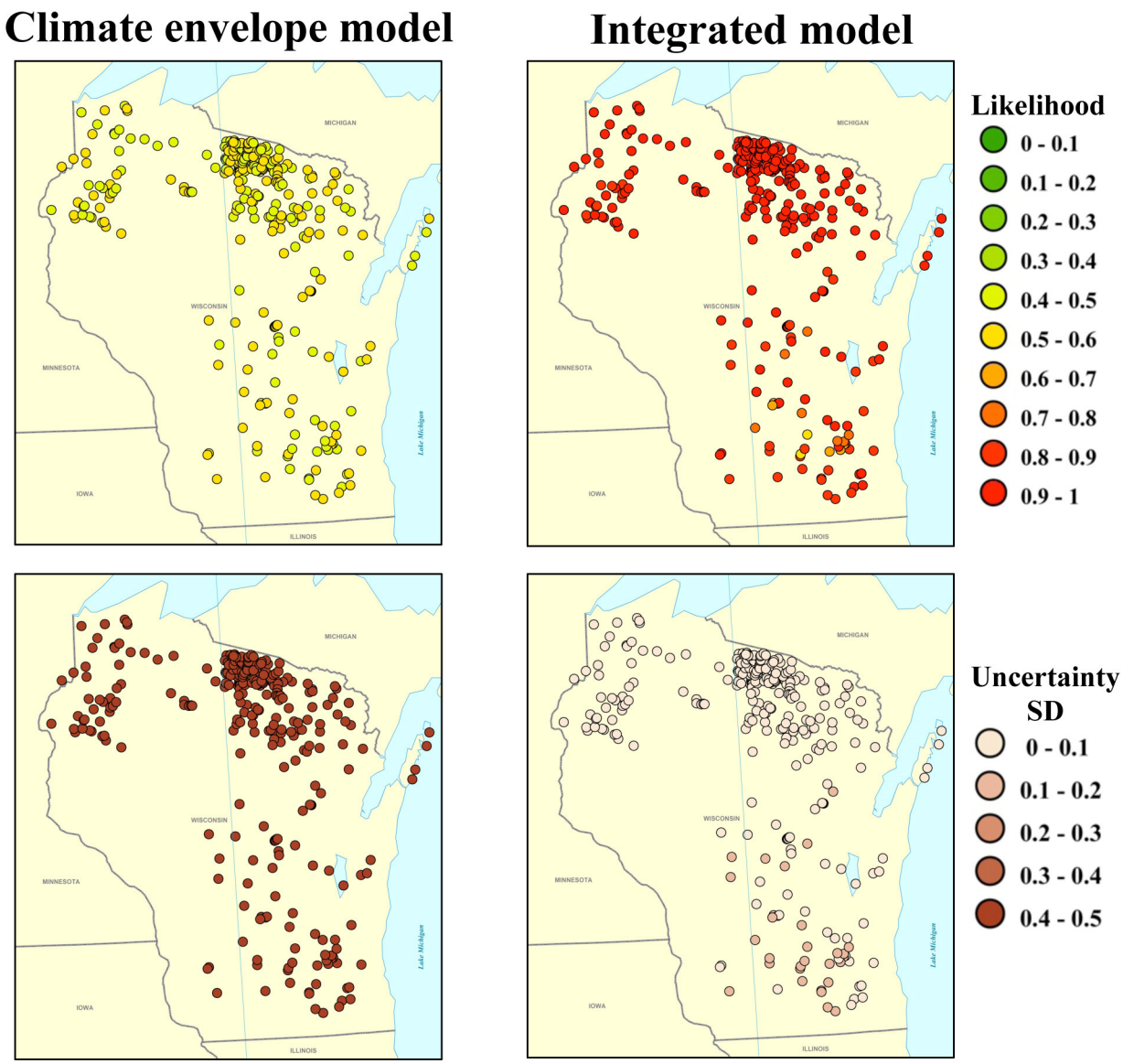

FIG. A6. *Orconectes* colonization probabilities for 292 Wisconsin lakes from the climate envelope colonization (left panels) and integrated (right panels) models. Null model DIC: 405, Integrated model DIC: 324.

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